U.S. DEPARTMENT OF COMMERCE
National Technical Information Service

PB-272 287

Identification and Specification of Inputs for Benefit-Cost Modeling of Pesticide Use

Pennsylvania State Univ, University Park

Prepared for

Environmental Protection Agency, Washington, D C
Point Source Management Div

Aug 77

PB 272 287

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U.S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA. 22161

TECHNICAL REPORT DATA
(Please read Instructions on the reverse before completing)

1. REPORT NO. EPA-600/5-77-012	2.	3. RECIPIENT'S ACCESSION NO. PB 272 287
4. TITLE AND SUBTITLE IDENTIFICATION AND SPECIFICATION OF INPUTS FOR BENEFIT-COST MODELING OF PESTICIDE USE		5. REPORT DATE August 1977 issuing date
		6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S) D. J. Epp, F. R. Tellefsen, G. A. Shute, R. M. Bear, K. P. Wilkinson		8. PERFORMING ORGANIZATION REPORT NO.
9. PERFORMING ORGANIZATION NAME AND ADDRESS The Pennsylvania State University University Park, Pennsylvania 16802		10. PROGRAM ELEMENT NO. 1HB617
		11. CONTRACT/GRANT NO. R863247-01-1
12. SPONSORING AGENCY NAME AND ADDRESS Office of Air, Land and Water Use - Wash., DC Office of Research and Development U.S. Environmental Protection Agency Washington, DC 20460		13. TYPE OF REPORT AND PERIOD COVERED Final
		14. SPONSORING AGENCY CODE EPA/600/16

15. SUPPLEMENTARY NOTES

16. ABSTRACT

Benefit-Cost (B/C) analysis requires inputs which are inclusive, valid, quantifiable and reliable. Proper attention to the procedures outlined in the six sections of this report will give a broad, although not necessarily complete, review of the effect of a pesticide-use decision. The six major sections are:

(1) A taxonomy of pesticide use effects. (2) Economic production -- Improvement in production is the major benefit of pesticide use. Although alternative cost assumptions are commonly used in B/C modeling, the opportunity cost approach is argued to be a superior procedure. (3) Human health -- Statistical methods are identified to measure the health costs of pesticide use. Chronic health effects are the most difficult to analyze but methods are shown for measuring both dollar and utility values. (4) Environmental impact -- Impacted organism/effects matrices are developed to insure comprehensive consideration of environmental impacts even though the state-of-the-art does not permit direct incorporation of these effects into the B/C ratio. (5) Aesthetic impacts -- Organism/effects matrices are developed for aesthetic impact assessment. (6) Distribution effects -- Social class, income class and international trade effects are presented as important considerations exogeneous to the B/C ratio.

A bibliography is presented with each major section.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Pesticide Economic Analysis Benefit/Cost Analysis Aesthetics Equity Econometric Statistical Analysis	Environmental Economics Environmental Management Human Health Effects Enviroimental Impacts Risk/Benefit Analysis	05A 05C 13B

18. DISTRIBUTION STATEMENT UNLIMITED	19. SECURITY CLASS (This Report) UNCLASSIFIED	21. NO. OF PAGES 157
	20. SECURITY CLASS (This page) UNCLASSIFIED	22. PRICE PC A08 MF A01

EPA Form 2220-1 (9-73)

EPA-600/5-77-012
August 1977

IDENTIFICATION AND SPECIFICATION OF INPUTS FOR BENEFIT-COST MODELING OF PESTICIDE USE

by

Donald J. Epp
F. Roger Tellefsen
Gary A. Shute
Robert M. Bear
Kenneth P. Wilkinson
The Pennsylvania State University
University Park, Pennsylvania 16802

Grant No. R803247-01-1

Project Officer

Darwin R. Wright
Agriculture and Non-Point Source Management Division
Office of Air, Land, and Water Use
Washington, D.C. 20460

OFFICE OF AIR, LAND, AND WATER USE
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

DISCLAIMER

This report has been reviewed by the Office of Air, Land and Water Use, U. S. Environmental Protection Agency, and approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of the U. S. Environmental Protection Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

ABSTRACT

Benefit-cost analysis requires inputs which are inclusive, valid, quantifiable, and reliable. Unfortunately, much of the data presently available on pesticide use does not measure up to these qualifications. The purpose of this report is to define, delineate, and improve data inputs to benefit-cost analysis. Specific objectives established to support this purpose are:

- to provide a comprehensive classification scheme incorporating all impacts resulting from a favorable pesticide-use decision;
- to establish conceptually valid approaches to the quantifications of major effects of pesticide use;
- to identify and usefully display areas of inputs where quantification is currently inappropriate and, if attempted, could produce misleading or seriously biased results;
- to illustrate the application of benefit-cost analysis through examples based on the use of aldrin on corn.

Proper attention to the six sections of the procedure developed in this report will give a broad, although not necessarily complete, review of the effects of a pesticide-use decision and will greatly assist comparison of the benefits and costs. The six major sections are:

- A taxonomy of pesticide use effects -- This section provides an overview of pesticide-use decision variables.
- Economic production -- Improvement in economic production is the major benefit of pesticide use. This section identifies competing means of evaluating production effects. Although alternative cost assumptions are commonly used in benefit-cost modeling, the opportunity cost approach is argued to be the only concestually valid technique and a generally superior procedure.
- Human health -- This section identifies statistical methods employed in the measurement of health costs resulting from pesticide use. Because chronic health effects are the most difficult to analyze, an emphasis is placed on evaluation techniques for measuring costs of chronic illness which are shown to be measurable in both dollar and utility values.
- Environmental impacts -- Relevant factors here and in the next sections are not generally quantifiable to a benefit-cost framework. They remain, however, as important considerations to pesticide-use decision making

and Impacted Organism/Effects matrices are developed as a tool to insure comprehensive consideration of environmental impacts.

- Aesthetic impacts -- This section provides a process for aesthetic impact assessment through the use of Organism/Effects matrices.
- Distribution effects -- Like environmental and aesthetic impacts, distribution effects are treated as exogenous to a benefit-cost scheme. This final section considers the distributional effects of pesticide use when stratified by social, income and international trade classifications.

CONTENTS

Page

ABSTRACT iii

FIGURES viii

TABLES ix

ACKNOWLEDGEMENTS x

CHAPTER 1: INTRODUCTION AND FINDINGS 1
- The Nature and Use of Chemical Pesticides 1
- Purpose of Investigation 1
- Regulation of the Pesticide-Use Decision 1
- Benefit-Cost Procedures 2
  - Use of money as a common unit of measure 3
- Objectives and Limitations of Study 4
- Outline of Subsequent Sections 5
- Major Findings and Conclusions 6

CHAPTER 2: A TAXONOMY OF PESTICIDE USE EFFECTS 7
- The Form of Taxonomy 7
- Human and Non-Human Health Effects 7
- Agricultural Impact of Pesticide Use 11
- The Effects of Pesticide Use on Materials and Property 13
- Human Health Impact Analysis 13
- Environmental and Aesthetic Impact Analysis 13
- Regulatory Control Costs 17
- Distributional Aspects of Pesticide-Use Decisions 17

CHAPTER 3: ECONOMIC PRODUCTION 18
- The Nature of Agricultural Production in the United States 19
- General Characteristics of the Agricultural Production Model 19
- Pesticide Output Information From An Agriculture Production Model 20
- Methods of Analyzing Pesticide-Use Effects 21
  - The alternative cost approach 21
  - The opportunity cost approach 23
- Comparison of Methods 25

CONTENTS (Continued)

Page

Choosing Between Alternative Cost and Opportunity Cost Methodologies 27
Additional Information Needs 29
Illustrations From Aldrin Use on Corn: A Case Example 30
    The model 30
    Model results 32
    Weakness of the model 39
Summary and Comment on Recent Literature 40
Bibliography for Economic Production Chapter 43

CHAPTER 4: HUMAN HEALTH 47

Review of Literature 47
The Human Health Model 48
Measuring Pesticide Related Illness 53
    Acute injury 53
    Chronic injury 54
Techniques for Evaluating Risk Levels 55
    "No positive result" method 55
    Mathematical extrapolation 56
Techniques for Evaluating a Change in Mortality Rate 62
    Foregone earnings approach 63
    Other approaches 65
    Inadequacy of evaluation techniques 65
    Willingness to pay 66
Human Health Effects of Corn on Aldrin: A Case Study Example 72
Bibliography for Human Health Chapter 81

CHAPTER 5: ENVIRONMENTAL IMPACTS 86

Introduction 86
General Review of Pesticide Movement and Problems 87
    Insects 88
    Birds 88
Effects of Pesticides on the Terrestrial Ecosystem 90
    Soil invertebrates 90
    Mammals 91
Fresh Water Aquatic Effects 91
Environmental Impact Assessment 92
    Importance of species survival 92
    Measurement of environmental effects 93
    The food chain model 95
    The model 95
    DDT example 101
    The impact matrix 103
    The Delphi technique 106

CONTENTS (Continued)

Page

Environmental Impacts of Aldrin Applied to Corn:
  A Case Study Example 106
    Soil complex 107
    Higher terrestrial trophic levels 108
    Aquatic sector 110
    Summary 111
Aldrin-Dieldrin Impact Matrix 113
    Matrix operation 113
    Summary 117
Bibliography for Environmental Impacts Chapter 119

CHAPTER 6: AESTHETIC EFFECTS 127

Problems of Quantification 128
Aesthetic Impact Assessment 129
The Aldrin Case 131
Bibliography for Aesthetics Effects Chapter 132

CHAPTER 7: DISTRIBUTION EFFECTS 133

Geographic Distribution of Effects 133
Social Effects 134
    A social accounting system 135
    Community level assessment 137
Balance of Payment Effects 138
    Impact of foreign trade 139
    The affect on the U. S. trade balance
      of restricting aldrin on corn 142
Bibliography for Distribution Effects Chapter 143

FIGURES

Number		Page
1	Health effects and impact matrix	8
2	Analysis of a shift in supply	24
3	Consumption regions	31
4	Change in the number of acres of corn by region	36
5	Change in corn production by region	38
6	Option demand	49
7	Human health assessment model	50
8	Dose-risk extrapolation using Mantel-Bryan procedure	59
9	Effect of changing confidence levels in Mantel-Bryan procedure	61
10	Safety indifference map	68
11	General pesticide movement model in faunal sub-system	96
12	Pesticide movement in the total system	98
13	General impact matrix	104
14	Concentration of dieldrin in marine organisms	112
15	Aldrin-dieldrin impact matrix	114
16	Pesticide impacts on activities with aesthetic components	130

TABLES

Number Page

1 Summary of Health Effects 9
2 Summary of Impacts 10
3 Agricultural Impacts of Pesticide Decisions 12
4 Material and Property Damage Impacts of Pesticide Decisions 14
5 Human Health Impacts of Pesticide Decisions 15
6 Natural Environment and Aesthetic Impacts of Pesticide Decisions 16
7 Change in the Number of Acres in Specified Crops by Region 33
8 Change in Production of Specified Crops by Region 35
9 Change in Acres of Corn in Each Soil Productivity Group by Region 37
10 Extrapolated Values of "Safe" Doses for Three Different Dose-Response Curves Describing Observed Responses in 2% to 50% Response Range Equally Well 58
11 Fraction of Experimental Dose Using Probit Extrapolation with Different Slopes for an Estimated Risk of 1/100,000,000 60
12 Dose-Risk Relationships for Dietary Dieldrin Using the Mantel-Bryan Extrapolation Technique 74
13 Relative Composition of Diet by Food Class 75
14 Residues of Dieldrin by Food Class 77
15 Major U. S. Export Crops (1974-1976) 140
16 United States Exports of Major Crops As A Percentage of Total United States Production 141

ACKNOWLEDGMENTS

Several people were particularly helpful to the authors at various stages in the preparation of this work. Dr. Michael J. Hay was the EPA Project Officer during most of the period of the grant and provided valuable advice and assistance. Dr. Terry A. Ferrar, Director of the Center for the Study of Environmental Policy, The Pennsylvania State University, also provided generous assistance in solving unusual administrative problems which arose during the grant period.

Technical advice on pesticide effects and related biological science material was provided by Dr. Herbert Cole, Jr., Professor of plant pathology, The Pennsylvania State University, Dr. James W. Gillet, Terrestrial Effects Branch, Environmental Research Lab, EPA, and Dr. Frank M. Fisher, Professor of Biology, Rice University.

CHAPTER 1

INTRODUCTION AND FINDINGS

THE NATURE AND USE OF CHEMICAL PESTICIDES

The widespread use of chemical pesticides, be they insecticides, herbicides, fungicides, or rodenticides, has resulted in broad scientific and public concern about the effects of such usage. It is widely appreciated that pesticides play a critical role in the production of food in modern agricultural nations, contributing to hitherto unparalleled levels of productivity. In the home and in the community, pesticides can result in control of noisome, unpleasant, and sometimes dangerous pests.

It is no less widely appreciated that pesticides themselves may be dangerous. Many pesticides are generally believed to persist in the environment, and to biomagnify through the food chain with adverse or unknown consequences. Some pesticides are believed to be tumorogenic or carcinogenic. Some pesticides, in relatively small dosages, can cause acute human health problems. Some pesticides can cause adaptations of target species into more pesticide-resistent strains for which control is more difficult. Some pesticides may seriously damage non-target species.

PURPOSE OF INVESTIGATION

In the broadest sense, it is the purpose of this investigation to identify comprehensively both positive and negative consequences of pesticide use and to specify all impacts in a manner most conducive to the evaluation of pesticide effects. It is not the objective of this study to develop a full benefit-cost decision model, but only to extend knowledge of the relevant inputs to such a model.

REGULATION OF THE PESTICIDE-USE DECISION

The Federal Insecticide, Fungicide and Rodenticide Act (FIFRA), as amended and supplemented by the Federal Environmental Pesticide Control Act (FEPCA) of 1972, mandates the Administrator of the Environmental Protection Agency to exercise comprehensive regulatory control over the production, distribution, and use of pesticides. The fundamental regulatory authority accorded the Administrator is that requiring registration as a prerequisite to "distribute, sell, offer for sale, hold for sale, ship, deliver for shipment, or receive and (having so received) deliver or offer to deliver to any person . . ." (FIFRA, Section 3). As a condition of registration, the use (e.g., home use for flying insects) must be specified in the application for registration, along with the pesticide name, formula, pertinent test data, and a request that the pesticide be classified under the law for general use, restricted use, or both.

Classification is made at the discretion of the Administrator (subject to review) accordingly as he or she determines whether or not the pesticide, if used in accordance with instruction, will generally cause unreasonable adverse effects on the environment (including injury to the applicator). Where it is determined that such adverse effects would generally result, the pesticide is classified for restricted use and may be applied only by, or under the direct supervision of, a certified applicator in accordance with labeling instructions. If such effects are not expected, the pesticide is classified for general use and may be generally applied provided only that applications be in accordance with labeling instructions.

FIFRA and FEPCA, along with regulations promulgated thereunder, provide further for issuance of permits for experimental use of pesticides, for a procedure for cancellation of the registration of pesticides in some or all uses, for reclassification for general use and restricted use (or both), and for suspension of registration for some or all uses pending the outcome of cancellation proceedings where an imminent hazard exists.

The principal criterion in the law on which the Administrator is to base the pesticide-use decision is his assessment of the degree to which the pesticide usage in question would result in "unreasonable adverse effects on the environment." According to FIFRA, the meaning to be given this phrase is "any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of any pesticide" (FIFRA, Sec. 2(bb)). The Act is thus quite clear that the Administrator's decision is to take explicit account of costs and benefits (economic, social, and environmental).

BENEFIT-COST PROCEDURES

In developing an analytical procedure to take explicit account of costs and benefits, it is important to focus on the type of decision for which this procedure is most critical. Some chemicals presented for registration have no known harmful side effects. There is very little problem about registering these for general use. Chemicals of this type do not present a serious hazard even if the user misapplies them or incorrectly prepares the solution from the purchased items. While not advocating sloppy constitution or application of the pesticides, these chemicals do not present a significant hazard even if carelessly handled. A second group of chemicals are extremely dangerous. These require specialized equipment for handling and extreme caution and careful monitoring in order to insure that side effects are kept at a minimum. These chemicals are so hazardous that they can only be approved for experimental use. A third group of chemicals have some harmful side effects but are extremely useful (or potentially so) in controlling economically important pests. The registration decision for these chemicals requires a determination of whether the pest control effects (benefits) are sufficiently important to offset the potential harmful effects (costs) of using these chemicals. It is for this third group that benefit-cost analysis is required.

The ideal of the benefit-cost approach is to reduce all of the effects to a common denominator so that a ratio of benefits to costs for selected alternatives can be constructed and weighed. The decision rule for pesticide use would be:

$$\frac{\text{P.V. } [\Sigma \text{ Benefits (Pesticide XXX)}]}{\text{P.V. } [\Sigma \text{ Costs (Pesticide XXX)}]} > 1.0$$

where: XXX = pesticide use under regulatory review

P.V. = present value

Σ = summation of dollar amounts

The expression "greater than 1.0" recognizes that any situation where the present value of the benefits stream exceeds the present value of the costs stream is efficient.

In formulating a benefit-cost analysis, the analyst must exercise care that the appropriate alternatives are compared in computing benefits and costs. In pesticide-registration decisions this would include consideration of alternative degrees of restriction (e.g., general or restricted use classifications), alternative geographic restrictions (e.g., registered for use only in particular states), as well as alternative (substitute) pesticides. If an alternative pesticide is likely to be used if the one under consideration is not registered for use, the analyst must be certain that the production effects, human health effects and environmental effects are all included in the analysis. Once fully specified and calculated the decision criterion is clear: if the best ratio is greater than one, the pesticide is socially useful; if the ratio is less than one, the pesticide should not be registered for use.

Use of Money as a Common Unit of Measure

In this report the term benefits refers to the desirable effects which would result from a decision to register the pesticide under review. The term costs, on the other hand, refers to the undesirable effects that would result from the decision to register the pesticide. Either benefits or costs can be measured in terms other than dollars. For example, benefits from registering a particular pesticide might be measured in bushels of grain not destroyed by insects while the costs might be measured by the percentage reduction in the songbird population of the area. It is clear that comparison of benefits and costs is very difficult when each is measured in different units.

However, in practice it is extremely difficult to reduce all pesticide-use effects to a common unit of measure. Several sectors permit measurement in terms of money units. These can give a quantified benefit-cost ratio in dollar terms. Most prominent of these sectors is economic production where most, if not all, of the effects can be measured in dollar terms. Some human and non-human health effects can be measured in dollar terms but others cannot. This poses a question as to whether the effects that can be quantified should be included with a common benefit-cost ration or if all human health and non-human health effects should be retained as a separate (exogenous) analysis. We find that while many effects can be reduced to dollar terms, some cannot without risk of bias. Given the current state of the art, a benefit- cost procedure by itself is not all conclusive: separate consideration of exogeneous factors must be undertaken. This problem is addressed in some detail in the next section.

OBJECTIVES AND LIMITATIONS OF STUDY

Benefit-cost analysis requires data which is inclusive, valid, quantifiable, and reliable. Unfortunately, much of the data presently available on pesticide use does not measure up to these qualifications. The purpose of this report is to define, delineate, and improve data inputs to benefit-cost analysis. Specific objectives established to support this purpose are:

- to provide a comprehensive classification scheme incorporating all impacts resulting from a favorable pesticide use decision;
- to establish conceptually valid approaches to the quantification of major effects of pesticide use;
- to identify and usefully display areas of inputs where quantification is currently inappropriate and, if attempted, could produce misleading or seriously biased results;
- to illustrate the application of benefit-cost analysis through examples based on the use of aldrin on corn.

In dealing with econometric problems covering a broad range of modeling applications, it becomes necessary to limit analysis to obtain meaningful results. The major limitations accepted by this study are the following:

- Emphasis throughout is placed on agricultural-use decisions. Commercial, industrial and home use applications are not explicitly considered in depth.
- Although a complete taxonomy of effects is developed, only the most critical areas of benefits and costs are subjected to detailed analysis. For example, production benefits and health costs are emphasized but production costs and health benefits of pesticide use are not fully treated.

- As a result of the limited focus, a comprehensive benefit-cost example illustrated with a general case is inappropriate. Instead and where appropriate, the specific impact categories considered will be illustrated with partial case analyses.

- The issue of discounting future benefits and costs to present value is not considered. All flows are presumed to be discounted at a zero rate.*

OUTLINE OF SUBSEQUENT SECTIONS

To obtain our stated objectives, we have developed a procedure with the following six basic sections: (1) a taxonomy of pesticide use effects; (2) economic production; (3) human health; (4) environmental impacts; (5) aesthetic impacts; and (6) distribution effects. It is our feeling that proper attention to these six sections and subpoints developed under them will give a broad, although not necessarily complete, review of the effects of a pesticide-use decision and will greatly assist comparison of the benefits and costs. Most of the sections are relatively self-explanatory and may be briefly described as follows:

A Taxonomy of Pesticide Use Effects -- This section provides an overview of pesticide use decision variables.

Economic Production -- Improvement in economic production is the major benefit of pesticide use. This section identifies competing means of evaluating production effects. Although alternative cost assumptions are commonly used in benefit-cost modeling, the opportunity cost approach is argued to be a generally superior procedure.

Human Health -- This section identifies statistical methodology employed in the measurement of health costs resulting from pesticide use. Because chronic health effects are the most difficult to analyze, an emphasis is placed on these effects and evaluation techniques for measuring costs of chronic illness are presented. Acute health effects must also be included in actual analysis, but because the methods and concepts involved are straightforward, they are given only passing treatment in this report.

Environmental Impacts -- Relevant factors here and in the next sections are not generally quantifiable to a benefit-cost framework. They remain, however, as important considerations to pesticide-use decision making. This section develops Impacted Organism/Effects matrices as a tool to insure comprehensive consideration of environmental and aesthetic impacts.

---

*This assumption was made at the suggestion of the Environmental Protection Agency. While the assumption obviously begs some significant questions, it was agreed that discussion of these topics should be left to some other report.

Aesthetic Impacts -- This section provides a process for aesthetic impact assessment through the use of Organism/Effects matrices.

Distribution Effects -- Like environmental and aesthetic impacts, distribution effects are treated as exogenous to a benefit-cost scheme. This final section considers the distributional effects of pesticide use when stratified by social, income, and international trade classifications.

For this study we have developed procedures whereby most of the significant human health effects can be quantified in dollar terms suitable for inclusion with the economic production effects in a benefit-cost ratio. Because the current state-of-the-art does not permit many of the non-human health effects of significance to be quantified, we have chosen to maintain all of the effects on non-human species on a descriptive basis rather than incorporating portions of it into a benefit-cost ratio. The first two sections provide the basic core and all of the quantified benefit-cost ratio considerations for the study. The distribution sections and the section on costs of environmental and aesthetic impacts will be presented predominantly as displays of descriptive materials.

MAJOR FINDINGS AND CONCLUSIONS

Specific findings are found in each major section of the report. In summary, findings may be classified as related to one of the following three broad areas of outcomes and conclusions:

- Opportunity cost methodology is the only conceptually valid technique in benefit-cost modeling. Consumer surplus is the output of opportunity cost analysis and is the proper measure of both economic and health effects associated with pesticide use.
- Chronic health risks can be reduced to both dollar and utility values. The study reviews the recent development of a method for determining such values.
- The present state-of-the-art is not sufficient to adequately quantify value in the areas of environmental and aesthetic impacts. However, the treatment developed in this study takes the first steps toward a conceptually sound quantified measurement.

CHAPTER 2

A TAXONOMY OF PESTICIDE USE EFFECTS

To be certain of comprehensive coverage of the effects of a pesticide-use decision, the analytical procedure must include a taxonomy of effects. This taxonomy should include beneficial as well as harmful effects and must include all categories of effects which might be significant in a pesticide-use decision. Obviously, no single decision will require detailed analysis of every point in the taxonomy. Rather, the analyst will examine a given use decision in light of the taxonomy to select the effects relevant to the particular pesticide and use under consideration.

A taxonomy to display benefits and costs for a pesticide-use decision has been developed in a matrix format. It is believed that this format makes most clear the effects of a pesticide and foreshadows the organization of subsequent sections of this report which describe methods appropriate for use in analysis of the effects. It also assures that all relevant entities receiving benefits or incurring costs due to pesticide use are considered, and that all critical economic or environmental impacts are identified. An important contribution of the taxonomy approach is its assurance that no relevant impact is inadvertently excluded from consideration.

THE FORM OF TAXONOMY

Figure 1 outlines the major headings of the taxonomy, broadly divided into health effects and impacts. Table 1 divides the health effects into human and non-human components, while Table 2 divides the impacts into more specific subheadings. Subsequent tables give a more detailed breakdown of each impact identified in Table 2. Thus, to construct the complete taxonomy with all details included, it is only necessary to place Table 1 and Tables 3 to 6 in the appropriate spaces as noted in Figure 1.

HUMAN AND NON-HUMAN HEALTH EFFECTS

A pesticide is designed to kill an unwanted organism. A perfect pesticide would stop there and would have no effect on non-target organisms in the ecosystem. However, such perfect pesticides do not yet exist and presently available pesticides can and do have certain deleterious effects on non-target organisms. It is the lack of specificity in these pesticides which requires a careful examination of effects on all organisms which are affected by the pesticide in question. If an organism is affected by a pesticide but its economic or environmental impact is small, it can be

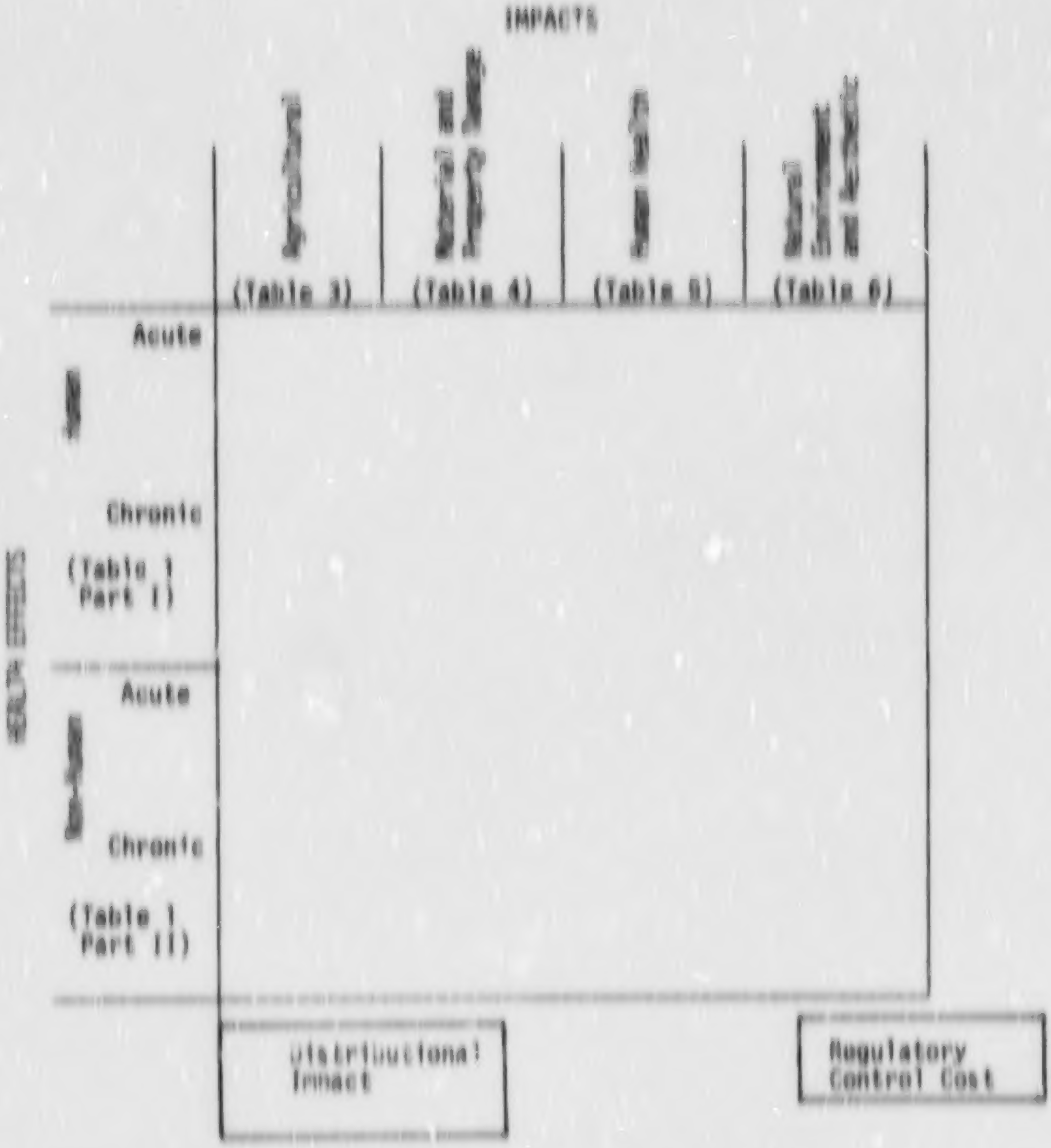

IMPACTS
Agricultural
Material and Property Damage
Human Health
Natural Environment and Aesthetic
(Table 3)
(Table 4)
(Table 5)
(Table 6)
HEALTH EFFECTS
Human
Acute
Chronic
(Table 1 Part I)
Non-Human
Acute
Chronic
(Table 1 Part II)
Distributional Impact
Regulatory Control Cost

Figure 1. Health effects and impacts matrix.

TABLE 1. SUMMARY OF HEALTH EFFECTS

I. Human

  A. Acute

    1. Death due to direct exposure
    2. Illness

  B. Chronic

    1. Mutagen
    2. Teratogen
    3. Oncogen
    4. Reproductive Impairment
    5. Chronic Physiologic Malfunction

II. Non-Human

  A. Acute

    1. Death due to direct exposure
    2. Contributory to death

  B. Chronic

    1. Mutagen
    2. Teratogen
    3. Oncogen
    4. Reproductive Impairment
    5. Chronic Physiologic Malfunction

TABLE 2. SUMMARY OF IMPACTS

I. Agricultural
   - A. Yield of crop
   - B. Quality of crop
   - C. Cost of production
   - D. Quality of land

II. Material and Property Damage
   - A. Right-of-way maintenance
   - B. Structural integrity of buildings
   - C. Damage to commodities during storage
   - D. Personal belongings

III. Human Health
   - A. Manufacturing worker
   - B. Formulator worker
   - C. Distributor - wholesale and retail
   - D. Applicators
   - E. Non-occupationally exposed
   - F. Disease vector control
   - G. Accident attenuation

IV. Environmental and Aesthetic
   - A. Non-renewable resources
   - B. Sporting activities
   - C. Tourism
   - D. Home and gardens

relegated to a footnote. The important point here is that the effect will have been considered, lessening the likelihood of a future "surprise." We have grouped this examination of species susceptibility under the general rubric of health effects and have broken these effects into human and non-human components (Table 1).

The human health and the non-human health components are organized in the same form, that is, division into acute and chronic effects. The major difference between the two health effects components is that the human health part includes a section on acute effects which do not lead to death while the non-human health part does not include such a section. This difference reflects the value position that sickness and discomfort in humans is an effect that should be considered in its own right. The same sickness and "discomfort" in plants and other animals is not of concern unless it contributes to the death of individuals. The non-human component is meant to be applied to animals and plants (be they terrestrial, fresh water, or esturine/marine) exposed to the pesticide under analysis. Since a pesticide can have varying effects depending on the organism, this component is further broken down into acute and chronic effects to allow a more thorough evaluation.

It is intended that the human and non-human health effects be iterative. This means that information about a non-human effect can be fed back into other non-human effects or into the human effects category where appropriate. This feature allows layers of effects to be worked through to their final impact, as in the case of biomagnification, while allowing each layer to be examined for its economic or environmental impact.

AGRICULTURAL IMPACT OF PESTICIDE USE

All crops produced for sale in the marketplace are considered as part of agricultural production. Thus, food crops such as wheat, corn, vegetables, and fruit, fiber crops such as cotton and timber, animal feed crops such as soybeans, oats, and hay as well as commercially produced flowers and house plants are included in this category.

The four considerations that are important in assessing the impact of a pesticide use decision in this area are change in crop yields, change in crop quality, change in cost of production, and change in land quality due to effects on soil microorganisms, soil erodability, etc. (Table 3). Collectively, these four considerations make it possible to predict the change in consumer surplus--the correct measure of benefit or cost.

For commodities which enter into international trade, it is not enough to measure the change in domestic consumer surplus. It is also important to obtain a measure of the change in foreign exchange earnings due to a pesticide-use decision. For commodities such as wheat and corn where a large percentage of the total output is exported, the balance of payments impact could be large. The measure of this distributional impact would be considered along with the change in domestic consumer surplus to evaluate the agricultural impact.

TABLE 3. AGRICULTURAL IMPACTS OF PESTICIDE DECISIONS

I. Yield - compared to best alternative method of pest control

II. Quality - compared to best alternative method of pest control

III. Cost of Production
    A. Machinery
    B. Land (quantity required)
    C. Fuel
    D. Labor
    E. Other inputs
    F. Constraints on production
        1. Spray intervals
        2. Re-entry intervals
        3. Cultural practices

IV. Land Quality
    A. Soil erosion
    B. Soil compaction
    C. Soil microorganisms

THE EFFECTS OF PESTICIDE USE ON MATERIALS AND PROPERTY

Included in this category are the commercial uses of maintaining rights-of-ways and maintaining the structural integrity of buildings and other man-made edificies as well as controlling damage to agricultural commodities during storage and controlling damage to personal belongings. An outline of these factors is presented in Table 4.

HUMAN HEALTH IMPACT ANALYSIS

Table 5 provides a complete list of groups who are likely to be exposed to pesticides and possibly suffer the adverse consequences or benefits listed under the human health effects category. Pesticide participants typically receive direct exposure to the pesticide through occupation or use and are most likely to be exposed to high concentrations of the chemical. As such, they are particularly prone to acute effects and should be the first group to experience any of the possible chronic health effects. Pesticide bystanders typically are exposed through residues, which are either ingested, inhaled or absorbed through the skin. It is in this type of exposure where the feedback feature of the health effects rows (Figure 1) permits a complete consideration of the problem. For example, pesticide run-off from crop land into streams may result in the magnification of pesticide residues as one moves up the food chain. At the top of the food chain, man may eat contaminated fish, become contaminated himself, and suffer health effects. Pesticide participants can also be pesticide bystanders within the definition of these terms. Obviously, a farmer can spray his fields and later eat contaminated fish from his stream.

Also included in the human health impact category are human health benefits from pesticide use in the form of disease vector control. Health benefits are also obtained from the reduction of human injury and death in accidents caused by road hazards visually obstructed by vegetation. Again, the health effects rows prevent any oversight in considering these benefits of pesticide use.

ENVIRONMENTAL AND AESTHETIC IMPACT ANALYSIS

In this category, we assess the impact of the pesticide on species which are not sold in a marketplace (Table 6). In many cases the impact will be negligible until a fairly high point on the food chain is reached, at which point the benefits or costs will be assessed. The most important impact will be in the case where a species is threatened with extinction due to health effects from a pesticide. Species extinction is an example of a loss of a non-renewable resource, and the impact will have to be carefully assessed.

The aesthetic impact deals with people's perception of their surroundings. Pesticide usage can materially affect this perception with the potential for either increasing or decreasing the perceived quality. Although

TABLE 4. MATERIAL AND PROPERTY DAMAGE IMPACTS OF PESTICIDE DECISIONS

I. Right-of-way Maintenance

  A. Damage to utility lines by vegetation

  B. Damage to property by fire along railroads and industrial sites

  C. Damage to property by motor vehicles due to visual obstruction of hazards by vegetation

II. Structural Integrity of Buildings

  A. Wood decay - railroad ties, etc.

  B. Insect damage - termites, carpenter ants, etc.

  C. Rodent damage - wiring, timbers, etc.

III. Damage of Agricultural Commodities During Storage

  A. Insects

  B. Fungus

  C. Rodents

IV. Personal Belongings

  A. Clothing - clothes moths

  B. Other belongings

TABLE 5. HUMAN HEALTH IMPACTS OF PESTICIDE DECISIONS

I. Pesticide Participants
  A. Manufacturing worker
  B. Formulator worker
  C. Distributors - wholesale and retail (container breakage)
  D. Applicators
    1. Farmer and farm employee
    2. Commercial applicators
      a. Industrial
      b. Home
      c. Farm
      d. Recreational
    3. Government workers
      a. Health control
      b. Right-of-way maintenance
      c. Large-scale insect control programs
  E. Government inspectors
II. Pesticide Bystanders
  A. Non-occupationally exposed
III. Disease Vector Control Benefits
IV. Accident Attenuation Benefits
  A. Human injury and death element

TABLE 6. NATURAL ENVIRONMENT AND AESTHETIC IMPACTS OF PESTICIDE DECISIONS

- I. Environmental
  - A. Non-renewable resources
    - 1. Terrestrial
      - a. Plant
      - b. Animal
    - 2. Fresh water aquatic
      - a. Plant
      - b. Animal
    - 3. Esturine/marine aquatic
      - a. Plant
      - b. Animal
- II. Aesthetics
  - A. Sporting activities
    - 1. Land based
      - a. Hunting
      - b. Camping and hiking
      - c. Picnicking
      - d. Bird watching
    - 2. Water based
      - a. Direct human contact
        - 1. Swimming
        - 2. Water skiing
        - 3. Other
      - b. Indirect human contact
        - 1. Fishing
        - 2. Boating
  - B. Tourism
  - C. Home and garden aesthetics

difficult to quantify, it is possible to at least identify groups within the society who are particularly concerned with aesthetic properties. Through their reaction to a change in the surroundings caused by pesticide usage, an estimate of the value placed on an aesthetic component by society as a whole can often be made.

REGULATORY CONTROL COSTS

Completeness requires that the costs of regulating pesticides be included in a benefit-cost analysis. This study is not a benefit-cost analysis of the entire pesticide control program, so it is only necessary to identify the control costs which are added if the pesticide is registered for restricted use. This value is added as a subtotal after the large matrix (Figure 1) is complete.

DISTRIBUTIONAL ASPECTS OF PESTICIDE-USE DECISIONS

Although this topic will not enter directly into a benefit-cost analysis, its consideration by the policy maker is imperative so that the effects of pesticide-use decisions on sectors as well as on the national economy are recognized. Some of the possible distributional effects might be a shift in agricultural production location causing unemployment in a geographic sector for a time, a change in the number or size of farms causing disruption of traditional land holding patterns, or a change in farm employment and employment trends (possibly a movement back to the farm, for example).

CHAPTER 3

ECONOMIC PRODUCTION*

One of the most important uses of chemical pesticides is to assist in the production of economically valuable goods and services. Many chemical compounds are used in the production of agricultural crops and livestock, forestry products, and for the protection of buildings and stored materials. By definition, a pesticide is a chemical compound used to protect these various crops and materials from damage by a particular pest such as insects, rodents, plants, and fungi.

Within this category of activities, the use of pesticides in the production of agricultural crops and livestock and forestry products has been a major consumer of chemical compounds. Many crops, such as cotton, require large quantities of chemical pesticides in order to be profitable in many areas of the country, whereas, for other crops, such as corn, the need for chemical pesticides depends upon the area. Some regions with many different pests that seriously affect corn production require large amounts of chemical pesticides; other areas of the country may require very small amounts of pesticide or perhaps no chemical compounds at all in order for corn production to be profitable. Thus, the amount and type of pesticide used in agricultural production and for forestry production varies greatly from one region to another. This variation in the use of pesticide is documented in the report by Andrilenas (2).**

Additional quantities of pesticide are used to protect materials that have been stored or to protect structures from damage by a variety of pests. These uses also vary from one region to another depending on the magnitude and degree of damage caused by various pests present in the regions and types of materials that are stored or the structures that need protection. For example, certain areas of the country are much more seriously infested with termites and require more chemical protection of structures from termite damage than do other regions of the country. Generally speaking, the use of chemical pesticides to protect structures and stored materials is a minor use of chemical pesticides. The majority of uses, and therefore the major concern over pesticide use, involves production of agricultural and forestry products.

---

*Participating researcher in this section is Principal Investigator Dr. Donald J. Epp, Associate Professor of Agricultural Economics, The Pennsylvania State University.

**Numbers in parentheses refer to items in the bibliography at the end of the chapter.

THE NATURE OF AGRICULTURAL PRODUCTION IN THE UNITED STATES

The foregoing indicates that one of the major problems in any benefit-cost analysis of a pesticide registration decision will be the determination of the benefits to be obtained from the use of that pesticide. The majority of cases requiring detailed benefit-cost analysis are likely to involve the use of chemical pesticides in the production of agricultural and forestry products. It is the purpose of this section to critically survey procedures for evaluating the contribution of pesticides to such production and to discuss the considerations involved in selecting a particular method appropriate to the decision at hand.

GENERAL CHARACTERISTICS OF THE AGRICULTURAL PRODUCTION MODEL

The production of agricultural crops in the United States takes place, for the largest part, in a complex and interconnected industry. Many of the inputs used to produce one commodity could be used to produce different crops. Most of the firms in this industry produce several to many different products and alter the proportions as production costs and product prices change. Due to a large public and private research effort, many aspects of agricultural production experience technological change. Frequently these changes are interrelated. Corn production provides a good example, for the hybrid varieties not only produce greater yields with a given set of other inputs, they also exhibit greater response to chemical fertilizers, show greater need for pest protection via chemical pesticides and respond to altered tillage and irrigation practices.

Because of the complex and interconnected nature of the agricultural industry it is not possible to accurately predict all of the likely consequences of altering production practices (such as pesticide use) using simple analytical approaches. More sophisticated analytical methods are needed to determine all of the important changes, some of which are not obvious. Models have been developed of sectors of the agricultural industry which permit refined analysis of changes within that sector.

To evaluate pesticide use in the production of agricultural crops, it is necessary to utilize a sophisticated model of United States agriculture. Such a model must be national in scope, recognizing that the markets for agricultural products and individual production regions are not isolated from one another. The model used must be regionalized to recognize the differences in the physical relationships which are a function of the location of agricultural production such as: soil type, crop yield, climate, pest infestation, and other factors

The model developed must permit the treatment of several crops as endogenous to the model. The use of several crops simultaneously recognizes that land use may be shifted from one crop to another depending upon the prices and the production cost. The model also needs to handle one or more livestock activities. It would be desirable that the model have activities relating to the production of grains and forage, although some portions of that sector can be handled with exogenous variables.

Another desirable feature of a national agricultural production model would be the ability to disaggregate by soil types to define various production activities on different types of soil. The production activities in such a model should be disaggregated by variable input so that labor, fertilizer, pesticides, machinery, and other relevant variables are specified and can be updated or changed as additional cost data becomes available.

The demand side of the model may be either developed by equation or specified exogenously. For purposes of a benefit-cost analysis, an endogenously determined demand for each of a set of demand regions would be best, but in some circumstances an exogenously specified demand may be necessary. Export demand should be explicitly considered and may be either added to the demand of a region from which it will be exported or assigned a regional location at a specific port.

In most cases the mathematical programming technique will be some form of linear programming model. This model should guarantee that all regional demands are satisfied, that no regional resource restraints are exceeded, and that the objective function is cost minimization, which seems most desirable for this purpose. Such a model will show the land use and production pattern emerging from the least cost solution which satisfied all constraints.

It is likely that a specific national model may need to be developed for evaluating pesticide decisions. It may be possible, however, to use existing models developed by the USDA or by other agencies, such as the models developed at Iowa State University as a part of the National Science Foundation--Research Applied to National Needs Program, or the agricultural model being developed by the Office of Pesticide Programs of EPA.

PESTICIDE OUTPUT INFORMATION FROM AN AGRICULTURAL PRODUCTION MODEL

The model solution will indicate the effect of a pesticide decision on the farm cost of producing not only the crop on which the pesticide is used, but also the effect on costs of producing related agricultural products. Assuming that markets for agricultural products are reasonably competitive, these changes in costs will be reflected in changes in the cost of consumer products. Thus, the production effect on consumer surplus of a pesticide can be calculated and compared with other consumer surplus effects of the pesticide decision. The actual conversion of farm price changes to consumer price changes involves the use of formulas developed by the U.S. Department of Agriculture. These formulas take into account the processing and transportation costs associated with the conversion of farm products to consumer products.

These changes in consumer prices, coupled with measures of consumer price elasticities for each product, permit calculation of the new equilibrium price and quantity for each affected consumer product. The equilibrium positions before and after the pesticide restriction can be compared to calculate a *change* in consumer surplus. (Note that total consumer surplus need not be calculated; only the change.) The change in consumer surplus is the final output of the economic production section and under procedures suggested in this report, is compared with similar changes calculated in the human health section for a final benefit-cost ratio.

The data needed for modifying the basic agricultural model is mainly that associated with agricultural production utilizing the pesticide under review. If this is a pesticide currently in use and being considered for withdrawal from registration, the agronomists, entomologists and livestock production specialists in those areas where the pesticide is used would be asked to estimate likely production practices and results if this pesticide were withdrawn from use. Consideration would be given to utilization of substitute pesticides and alternative production methods. If the pesticide is a new one being considered for initial registration, the same specialists would be consulted concerning the likely impact of this pesticide on agricultural production of the crop or crops for which it is intended to be registered. It is anticipated that these technical experts will base their judgment for new pesticides on the similarity to existing pesticides or the manufacturer's claims about the effects of the pesticide. A major difficulty in both cases will be correctly specifying the alternative practices to be considered if a pesticide is to be withdrawn. It may be possible to completely alter the production process and have very little impact on total output. On the other hand, it may be very difficult to completely anticipate the changes in production practices that must accompany a new pesticide. In all cases the analyst will need to identify the relevant experts in making the adjustment in production coefficients in the model.

METHODS OF ANALYZING PESTICIDE-USE EFFECTS

When analyzing agricultural production effects of a pesticide-use decision, there are two major alternative methods. These two are what we shall call the alternative cost method and the opportunity cost method. Either of these methods may employ models of the agricultural industry, but as will be shown in the following sections, the opportunity cost method requires more sophisticated models and delivers a more comprehensive description of likely effects.

The Alternative Cost Approach

With the alternative cost method, the analyst examines the additional cost necessary to achieve a given objective using the next best alternative method to the one under study. Applying this to the chemical pesticides decision, for example, would require that the analyst examine the next best method for producing one hundred bushels per acre of corn without the use of a particular insecticide. This approach requires that there be some alternative method available which will achieve the same level of output or degree of protection in order that the costs of the two alternative ways of producing the output can be compared.

In concept, the alternative cost approach of the prohibition decision for a particular pesticide in a particular use requires the following procedures.

- Identify the production function for the product using the pesticide under consideration, $z=h(x_1, \ldots, x_n)$, where z is the output and x_j ($j=1, 2, \ldots, n$) are inputs, one of which is the pesticide under consideration (x_n).
- Letting z_0 represent the output of the production process that would obtain under the status quo with pesticide use and C be the cost of producing that output under the status quo find:

  The minimum $c = \Sigma\, p_j\, x_j$

  such that: $h(x) > z_0$

  given $x_n = 0$.

  In this formulation, c gives the minimum cost of production assuming x_n cannot be used in this production process.
- Then $c - C$ is the alternative cost measure of the production benefits of using x_n in this production process.

The alternative cost approach has been used in several studies analyzing the effect of pesticide restrictions in crop production. Andrilenas (1) summarized three studies conducted by the U.S. Department of Agriculture and reported the costs to farmers of restrictions on three different pesticides or pesticide groups commonly used in agricultural production. All of the studies assumed that production was maintained at the level obtained prior to banning or restricting the pesticide. The studies then calculated the costs of such a ban on pesticide use. Typical of the results that can be obtained with this approach are the conclusions reported by Andrilenas:

> Banning the use of 43,000,000 pounds of phenoxy herbicides, primarily 2, 4-D and 2, 4, 5-T on 62,000,000 acres of crops would increase farmers' direct production costs about $290,000,000. This is about 1.5 percent of the farm value of all crops and 6.6 percent of farm value of crops treated . . . Besides these losses, farmers would use about 20,000,000 more hours of operator and family labor (1, p. 53).

Another study implying the alternative cost method was reported by Edwards (18). In his attempt to measure changes in external costs and benefits related to reductions in the use of chlorinated hydrocarbons, he assumed that the quality and yield of crops would be maintained by substituting organic phosphates for the restricted chlorinated hydrocarbons. Langham (33) discussed possible applications of the Edwards study to a national investigation of changes in external effects.

The Opportunity Cost Approach

A simplified partial analysis of the difference between the alternative cost method and the opportunity cost method is presented in Figure 2. The decision to ban a pesticide presently in use is illustrated by the shift in the supply curve. Curve S_0 represents the industry supply curve using the pesticide while curve S_1 represents supply under the higher cost next best alternative method of production which does not use the pesticide. The alternative cost method examines the increase in cost of obtaining the initial equilibrium quantity, Q_0. This increase in cost is represented by the area ABP_1P_0. The opportunity cost method, on the other hand, recognizes that a new equilibrium price and quantity will emerge after the shift in supply. Quantity will shift to Q_1 and price to P_2. Comparing the new equilibrium with the old, the analyst using the opportunity cost method observes that consumer surplus has been reduced by an amount equal to area ACP_2P_0 in Figure 2. This reduction in well-being is clearly less than the one calculated with the alternative cost method. Because the opportunity cost method recognizes changes in production and consumption, it is the preferred method. The alternative cost method overstates the cost of restricting pesticide use.

The above diagramatic analysis presents the essential features of the budgeting approach to the opportunity cost method. This is discussed in more detail below. Another approach, modeling, carries the above analysis further to show changes in the markets for factors of production as well as other markets. While this method cannot employ graphic analysis due to the complexity of relationships, the mathematical modeling techniques described and illustrated below permit a much more comprehensive analysis of likely effects of a pesticide-use decision.

As was noted above the opportunity cost approach measures the change resulting from the pesticide restriction as the change in consumer surplus in the economy. There is abundant literature discussing the problems of estimating consumer surplus; some authors even question the value of the concept in many empirical contexts. For the purpose of this exercise, however, it seems appropriate that consumer surplus be the criterion used. The ultimate consumer of the products and services produced with the pesticides should provide the basis for evaluation. This becomes particularly crucial where ramifications of the production shift may include a variety of products, not just the product using the pesticide as a direct input. Consumer surplus becomes the common denominator allowing comparison among a variety of different production effects. In essence, the opportunity cost approach calculates a shift in the supply curve for the product under consideration, calculates a new equilibrium quantity and price, and determines a change in consumer surplus by comparing the new equilibrium point with the old equilibrium point. As a practical matter, the opportunity cost approach may employ either of two methods: budgeting or modeling.

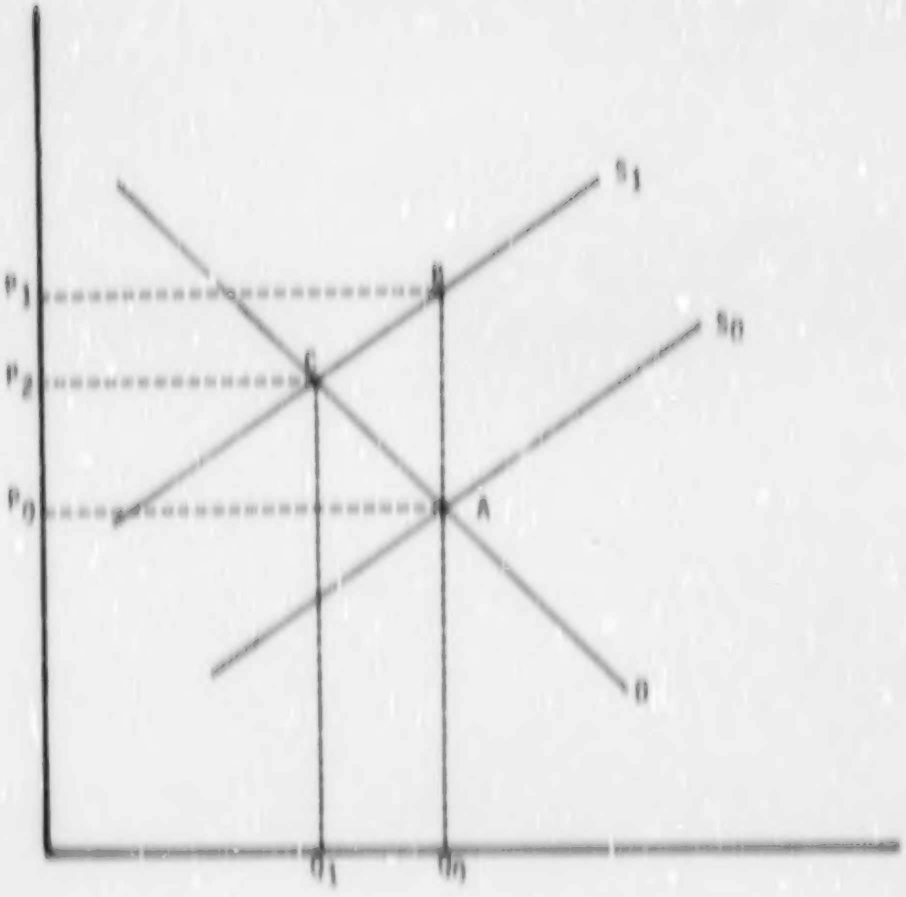

S_1
B
P_1
S_0
C
P_2
P_0
A
D
Q_1
Q_0

Figure 2. Analysis of a shift in supply.

Budgeting--

With budgeting, the analyst acknowledges that production shifts are likely for those products which are produced using the pesticide input which will now be restricted. Budgeting assumes that other commodities will be produced in the same amounts and at the same prices as with the status quo. Likewise, the budgeting approach has great difficulty in adjusting the factor prices of other inputs into the production of the products being considered. The budgeting approach is a more limited and restricted approach than the modeling one, but for some circumstances may prove advantageous.

Modeling--

The second and more complete method of applying the opportunity cost approach is through the use of production models. With this method, mathematical models of the economy (more likely the relevant portions of the total economy, such as the agricultural sector) are employed to trace the shifts in supply curves and the changes in the amount and price of various commodities and factors which result from the restriction on the use of a particular pesticide in the production of a particular crop (e.g., 11, 39). If these models are properly specified to include geographic areas and the various alternative production activities which take place in each of these areas, they are able to project changes in the location of production of particular crops and changes in the use of various production factors in each region of the country. This more realistically describes the likely reactions to the change in pesticide use and permits the calculation of a more accurate estimate of the change in consumer welfare resulting from the pesticide decision.

A few studies have attempted to employ an opportunity cost approach to examining pesticide restriction decisions. Casey and Lacewell (9) examined some of the effects of pesticide withdrawals on cotton production. They employed a limited regional model of cotton production to examine the likely impact of restricting herbicides and insecticides. The model, however, permitted only a limited number of substitute activities for cotton production and was able to only approximate the likely impacts from this change. Also, the authors did not follow through the changes in production costs to estimate an impact on consumer surplus.

Additional work has been done in the U.S. Environmental Protection Agency Office of Pesticide Programs (41). Much of this work has incorporated national production models on a regional basis using sub-state regions as the units of production and multi-state regions as the consuming areas. A variety of crops are permitted as alternatives in the models developed, and facilitate a more detailed examination of likely production shifts. Again, the production cost changes have not been translated into consumer surplus changes in this study.

COMPARISON OF METHODS

The alternative cost and opportunity cost methods are not equally adapted to handling all problems--each has several advantages and disadvantages. The alternative cost approach has the advantage of using data that is more easily obtained and not requiring the extensive development of mathematical models prior to the analysis. Since the alternative cost approach compares the status quo with the next best alternative, subject to a pesticide restriction, the only additional data needed beyond present production techniques is an estimate of how the same amount of product could be produced under the best alternative available with pesticide restrictions. This data can usually be obtained either from material supplied with the application for registration or from a few experts in the production of the crop under consideration. In addition, the alternative cost approach is usually less expensive since production adjustments are not considered, thereby eliminating the need for expensive models to incorporate changes in the use of factors of production and adjustment in production enterprises.

The major disadvantage of the alternative cost approach is that it is *conceptually erroneous*. Our knowledge of economic adjustments to changing production technology recognizes that adjustments will be made in the enterprise combinations and in factor combinations for producing a particular crop. If a significant factor of production is removed from use, the ensuing production adjustments will not be limited only to factors of production but may also include changing the crops produced or the location of crop production. In other words, restricting the use of a particular factor of production changes not only the production function but also the regional comparative advantage of various crops. The alternative cost approach ignores all of this knowledge by assuming that the crop in question, using the next best alternative, will be produced at the same level as before.

This conceptual error leads to a second error--the overstatement of the costs associated with pesticide restriction. Because this approach ignores the adjustments in the amount of agricultural product produced, it leads the analyst to a cost of production figure that is greater than the one that would actually result. As production costs increase it is likely that the quantity produced will decrease. Thus, the alternative cost approach has the analyst multiply a higher cost per unit by more units than would actually be produced. This leads to an erroneous calculation of the change in consumer surplus and the overstatement of the cost of making the pesticide restriction decision.

The major advantage of the opportunity cost approach is conceptual: It allows the analyst to see the adjustments that the economy is likely to make in response to changing production costs from a specific pesticide-use decision. The use of an econometric model of the agricultural sector, particularly, facilitates the analysis of changing comparative advantage and the resulting production pattern. This gives a more accurate indication of the ultimate cost of making the pesticide-use decision.

The use of models also can facilitate distributional analysis if these models have spatial or regional variables introduced into them. Through the use of regional supply and demand models, the analyst is able to estimate not only the market equilibrium supply and demand adjustments but also the regional production adjustments in response to the overall market changes. It is thus possible to note changes in the regional location of production (an adjustment that is likely to take place with pesticide restrictions) and to estimate the impact of these changes on various income and social groups that are distributed differently in the various regions of the country. Thus, the models that are used for the opportunity cost approach can facilitate a more detailed and sophisticated analysis of the effects of a pesticide-use decision.

The major disadvantage of the opportunity cost approach is that it requires a very large amount of data. Production costs, outputs, factor requirements, factor prices and a variety of other bits of information are required for each alternative crop activity in each region including alternative ways of producing the crop under consideration. Even if the supply equations and activities are limited to those most likely to enter into the solution, the requirements are formidable for most agricultural commodities. The demand side also requires regional consideration with specification of the demand for each of the commodities included in the supply side of the model. Careful attention must be given to the inclusion of complements and substitutes so that the model will give a reasonable approximation of actual market adjustments.

A second disadvantage of the opportunity cost approach is that the models developed for analyzing production and market shifts are usually short-run and static. This means that these models must be revised periodically in order to include new developments in factor prices, product prices and production technology. Thus, the models are expensive to maintain. They are also expensive to create in the first place. Most models require a great deal of prior research on the technical relationships in production and specification of the factors related to consumption. While much of this work has been done for agricultural commodities, it is recognized that the material is frequently inexact and often the models are out of date. It would be necessary, therefore, to undertake some rather expensive research in order to incorporate the opportunity cost approach for a commodity that did not already have substantial prior work.

Obviously, the detailed knowledge of production relationships and therefore the expense is reduced if one uses the budgeting method rather than the modelling method in the opportunity cost approach. Budgets usually involve a partial analysis of the adjustments and therefore do not require the development of production relationships for commodities not closely related to the commodity under consideration. Even so, the budgeting approach requires more information than the alternative cost approach because of the consideration of production changes. It is unlikely that the budgeting approach can be used for a regionalization analysis that involves more than a few regions. Thus, the lesser cost is somewhat offset by less information.

CHOOSING BETWEEN ALTERNATIVE COST AND OPPORTUNITY COST METHODOLOGIES

Since the two approaches, alternative cost and opportunity cost, differ in the amount of data that they require and in the cost of acquiring and processing that data, it is necessary for an analyst to choose between these two methods. Several points should be considered when making a selection of the most appropriate analytical method:

Demand elasticity--If the product for which the pesticide use decision is being made has a very inelastic demand, it may be appropriate to use the alternative cost approach. With a very inelastic demand, it is likely that the assumption of producing the same quantity regardless of cost is a reasonably close approximation to reality. Thus, the disadvantages enumerated above for the alternative cost approach are of much less significance for many agricultural products, such as vegetables, potatoes, cereals for human consumption and dry beans and peas.

Information availability--Since the opportunity cost approach requires a great deal of information about production relationships, not only for the commodity under consideration but for alternative commodities, it is helpful to have these relationships previously developed. Much of this information is so time consuming to develop that it would be virtually impossible to create a research program that could give results in time to be useful for any pesticide decision if a great deal of groundwork had not been laid prior to that research effort. If no information is available concerning production of the product in question or the alternative product that might be produced, it is likely that the alternative cost approach would be adopted. This approach requires less data and the material that it requires probably can be produced in a time frame that is reasonable for a pesticide-use decision.

Complexity of interrelationships--If the product is produced in many areas with a large variety of production alternatives, both with regard to factors of production and to alternative crops, one needs to use an opportunity cost approach. In such a situation the interrelationships are so complex that it is difficult to judge from casual observation of the data the likely combination that will result from a pesticide-use restriction. Under these conditions it is very desirable to use the opportunity cost approach if at all possible.

Availability of mathematical models--The stage of development of the mathematical programming models for a particular product or sector of the economy is an important consideration in choosing a method. If programming models are fairly well developed for most of the important alternative commodities as well as the commodity under consideration, it may be possible to modify the existing work relatively inexpensively to obtain the information needed on the pesticide-use decision. For example, a great deal of work has been done with the feed grain-food sectors of American agriculture. Less work has been done on the livestock sectors, although there are some models that incorporate feed grains and livestock. Pesticide-use decisions involving feed or food grains should give serious consideration to using an opportunity cost approach and some of the models that have been developed with appropriate modification. On the other hand, there has been very little modelling that includes speciality crops such as fruit and vegetables into a general agricultural model. It would be rather expensive and of dubious value to develop a programming model for pesticide use decisions involving those crops.

It is readily apparent that the key assumption of the alternative cost approach--no change in the quantity of product produced after restricting pesticide use--is not realistic in most cases and can lead to substantial error in estimating the social costs of a decision to restrict pesticide use. It is the opinion of the authors that the alternative cost approach is appropriate only in those cases where a substitute pesticide is readily available at virtually the same effective price as the pesticide under consideration or where alternative production techniques are available at no increase in per unit cost of production and which would involve not more than negligible shifts in the location of production. These are very restrictive conditions. For most pesticide-use decisions involving major agricultural crops these conditions do not hold and the opportunity cost approach is strongly advised.

ADDITIONAL INFORMATION NEEDS

Aside from the information concerning models and production relationships described above, the analysis of pesticide-use decisions requires additional specific information both on the crop production side and on the commodity demand side. As should be apparent from the preceding discussion the alternative cost approach needs only a limited amount of information compared to the opportunity cost approach. This section outlines the data needs for the opportunity cost approach, which includes all needs of the alternative cost approach.

Crop production data are needed for each region that produces a particular crop. Data needed include the inputs used, such as labor, machinery, gasoline, oil, fertilizer, chemical pesticides and other inputs, and the productivity of each of those inputs. This can best be summarized by stating the need for an empirically estimated production function for each crop, including alternative ways of producing the same crop for each region where this crop might be produced. In addition to the production function, the supply estimation side of the problem will require data on factor prices for each region. All of this information must be provided not only for the crop under consideration in the pesticide-use decision but for all alternative crops that might appear in any of the regions where the primary crop is produced.

The demand side of a model to be used in the opportunity cost approach is less restrictive concerning the kinds of data that must be provided. It would be desirable that a demand model be available and data be provided for each variable entering into the demand such as disposable income, age, ethnic composition of population in each of the consumption regions. If such a model or such data are not available, it is possible to use the opportunity cost approach employing an estimate of the quantity demanded for each commodity involved in the production side of the analysis. Such an approach assumes that changes resulting from production shifts will not significantly affect the demand for various commodities. For most products under consideration, this assumption is reasonably accurate.

ILLUSTRATIONS FROM ALDRIN USE ON CORN: A CASE EXAMPLE

Some of the advantages and difficulties associated with the modeling method of the opportunity cost approach that were discussed above can be illustrated using the registration decision of aldrin for use on corn. This is an example of a registration decision that has actually been considered by the EPA (41). In the course of its review, the EPA developed a mathematical model of the agricultural sector to analyze some of the production effects surrounding a decision not to permit chlorinated hydrocarbon pesticides to be used on corn. The model developed by the EPA included many of the desirable characteristics discussed above and illustrates the type of information that can be obtained from such an analysis as well as illustrating some of the problems with data and model specification.

The Model

The basic linear programming model contains activities which include crop production disaggregated by soil type and region, commodity transportation, conversion of commodities into feed nutrients, and conversion of feed grains into corn equivalents. Resource constraints are defined for land by land class. Demands include those for specific commodities, livestock nutrients, feed grains for export, and specific commodity exports.

Seven crops are included in the activities of the model--barley, corn, cotton, soybeans, oats, sorghum, and wheat. Other agricultural land uses are projected exogenously and subtracted from the total land base. The livestock sector is also projected exogenously and fixed at a predetermined level. The nonfeed grain portion of total livestock nutrient demands as well as the nonfeeding demands for endogenous crops (such as barley malt for brewing or corn for breakfast cereals) are projected using historical trends and population estimates. These are added to regional demands which must be met through local production or import from other regions.

The United States is partitioned into 129 producing areas, each of which is an aggregation of counties wholly contained within one state except for New England which is considered one producing region. The production areas are defined so as to have similar cropping conditions throughout each area.

Because of certain features of the model, it is most meaningful to present production results by consuming region. Therefore, production changes in the producing regions are aggregated into the 27 consuming regions composed of contiguous producing areas (Figure 3). In most cases, the consuming regions are whole states or aggregates of whole states. State boundaries are not observed, however, in those cases where commodity markets do not follow them. Where a state is split, it is because each part of the state lies in a separate market area.

Even though livestock feeding requirements are determined exogenously in this model, they are specified in terms of total digestible nutrients and crude protein. This allows the model to substitute various grains to satisfy these requirements. Likewise, the export requirement is partly specified exogenously for specific grains and partly specified in corn equivalents.

This allows the model to substitute feed grains as would be done in practice by livestock feeders in export markets.

The model is solved through a mathematical algorithm which simultaneously guarantees that: (a) all regional demands are satisfied, (b) no regional resource constraint is exceeded, and (c) the objective function is minimized. In so doing, the model solution depicts the land use and production pattern which satisfies all constraints at least cost.

The model was applied to the analysis of several alternative pesticide situations to determine the production effects and the impact on farm income by region of these alternative decisions. A first run of the model permitted chlorinated hydrocarbon insecticides to be used on corn in their historic amounts and areas. This was followed by a run which prohibited the use of any chlorinated hydrocarbons on corn. A particular situation is very similar to the one chosen for analysis in this illustration, that being prohibiting the use of aldrin on corn with no other chlorinated hydrocarbons available as a substitute. This second run did permit the use of organophosphate and carbamate pesticides as a substitute for the chlorinated hydrocarbon pesticides. Yields under the various pesticide assumptions were included in the model that are based on previous studies in the U.S. Department of Agriculture and the EPA relating to the effect of pesticide withdrawal and probable insect infestation.

Recognizing that farmers do not immediately respond to changing price in production conditions, the model included a flexibility constraint in the form of a flexibility charge. The flexibility charges impose an additional cost of production if production deviates beyond a specified percentage of historic trend for a given productive region.

The basic features of the EPA model are further elaborated by Arnold (3). One difference from the situation discussed by Arnold is that the present analysis did not assume yield impacts of restricting pesticide use which might occur with a moderate to heavy insect infestation. Rather, the analysis used for this illustration assumed a "most likely" or "typical" insect infestation with the corresponding adjustment in yield impacts of restricting chlorinated hydrocarbon pesticide use.

Model Results

On a national basis, the model shows an aggregate increase in planted corn acreage of about 88,000 acres. This results from the fact that corn is a more desirable feedgrain than alternative feedgrains in many areas and that farmers would increase their plantings slightly to offset minor yield reductions due to the restriction on chlorinated hydrocarbon pesticide (aldrin) use. Total changes in land used for each crop are reported in Table 7. Idle cropland (reported in the last column of Table 7) is reduced by almost 33,000 acres. This means that in total, 33,000 acres more cropland would be in production after a restriction on the use of chlorinated hydrocarbons than would be used if those pesticides remained available.

TABLE 7. CHANGE IN THE NUMBER OF ACRES IN SPECIFIED CROPS BY REGION

	Barley	Corn	Cotton	Soybeans	Oats	Wheat	Idle
			(Thousands of Acres)				
Boston							
New York							
Baltimore		0.53			-0.53		
Atlanta		140.48		-140.48			
Jacksonville							
Memphis							
Houston							
New Orleans							
Louisville							
Cincinnati							
Detroit							
Indianapolis							
Chicago	0.86	-84.84		84.84		-0.86	
St. Louis							
Des Moines							
Milwaukee							
Duluth		-38.45		6.62		13.11	5.61
Minneapolis		85.91			-0.27	-7.44	-54.48
Omaha							
Kansas City		-15.35					15.35
Amarillo			0.79			0.38	
Billings	2.07					-2.07	
Denver							
Salt Lake City							
Portland							
San Francisco							
Los Angeles			-0.22			-0.59	0.82
Total	2.94	88.29	0.57	-49.03	-0.80	1.74	-32.71
(Percent)	(0.03)	(0.16)	(0.01)	(-0.09)	(0.00)	(0.00)	(-0.21)

Grain sorghum: No change in any region.

The regional distribution of land use can also be seen in Table 7 and Figure 4. For example, the changes in acres planted to corn following the pesticide restriction vary greatly between regions. The largest reduction in corn acreage occurs in the Chicago region (84,840 acres) with lesser reductions in the Duluth and Kansas City regions. Offsetting these reductions in corn acreage are increases in the Atlanta region (140,480 acres) and the Minneapolis region (85,910 acres). Similar shifts can be observed for soybeans, barley, oats and wheat. Several instances can be found where land is shifted from one crop entirely into another. The Atlanta region is shown to shift 140,480 acres from soybean to corn production while the Chicago region shifts 84,840 acres from corn to soybeans. In other cases, the model shows where land is taken out of crop production. The Kansas City region shifts 15,350 acres from corn to idle while the Duluth region idles part (5,810 acres) of its 38,450 acre reduction in corn land. While the magnitude of these shifts is small in this example, it does illustrate the richness of detail that is possible from modeling approaches.

Changes in production in various regions may be even more important than changes in acres planted. Regional production changes for each crop are shown in Table 8 and Figure 5. While again, most of the changes are small relative to total production in the region and in total are a negligible change in national output, some regions do show fairly large production changes. The 11.9 million bushel reduction in corn production for the Chicago region might have a noticeable impact if it were concentrated in one part of the region. If distributed uniformly throughout the region, it is likely to have little effect since it only amounts to about one percent of total corn production. Likewise the 8.5 million bushel increase in corn production in the Minneapolis region and the 10.3 million increase in the Atlanta region only amount to 2.3 percent and 4.3 percent respectively of the corn production in those regions. Users of corn (livestock feeders and mixed feed manufacturers) likely will adjust to changes of this magnitude without major shifts in their location.

In some regions the model shows changes in the type of land used for crop production, even though the total acres devoted to a particular crop in the region remain constant. The land resource in this model was classified into three productivity groups, with group one soils the most productive and group three soils the least productive cropland. The change in acres planted to corn for each soil group in each region is shown in Table 9. In several regions (Memphis, Cincinnati, Detroit, Des Moines and Denver) the increases in corn acreage in one soil group are offset by decreases in corn acreage in other soil groups. In other regions the change in one soil group reflects the total regional change (for example, Atlanta or Chicago). The ability of models to illustrate changes in the types of soil used to produce each crop permit consideration of the location of production within the region and consideration of environmental impacts associated with production on different type soils. For example, the shift of nearly 200,000 acres of corn from group one soils to group two and three soils in the Des Moines region may result in greater erosion with accompanying sedimentation of lakes and streams.

TABLE 8. CHANGE IN PRODUCTION OF SPECIFIED CROPS BY REGION

	Barley (bu)	Corn (bu)	Cotton (bu)	Soybeans (bu)	Oats (bu)	Wheat (bu)
		(Thousands of Units)				
Boston						
New York						
Baltimore		40.41			-19.63	-0.13
Atlanta		10,289.38		-2,936.19		
Jacksonville						
Memphis				-0.07		
Houston						
New Orleans						
Louisville						
Cincinnati		114.70				
Detroit		-67.64				
Indianapolis		220.96				
Chicago	26.90	-11,889.55		2,833.08	31.99	-80.53
St. Louis		-1,281.46				
Des Moines		-1,736.74				
Milwaukee		-63.70				
Duluth		-2,126.51		126.40		415.92
Minneapolis	-0.10	8,533.24			-450.51	-221.75
Omaha		-38.07				
Kansas City		-1,805.99				
Amarillo			0.44			-7.51
Billings	40.28					-49.36
Denver						
Salt Lake City						
Portland						
San Francisco						
Los Los Angeles			-0.44			-56.63
Total	67.08	199.99	0.00	23.35	-438.15	
(Percent)	(0.02)	(0.00)	(0.00)	(0.00)	(-0.10)	

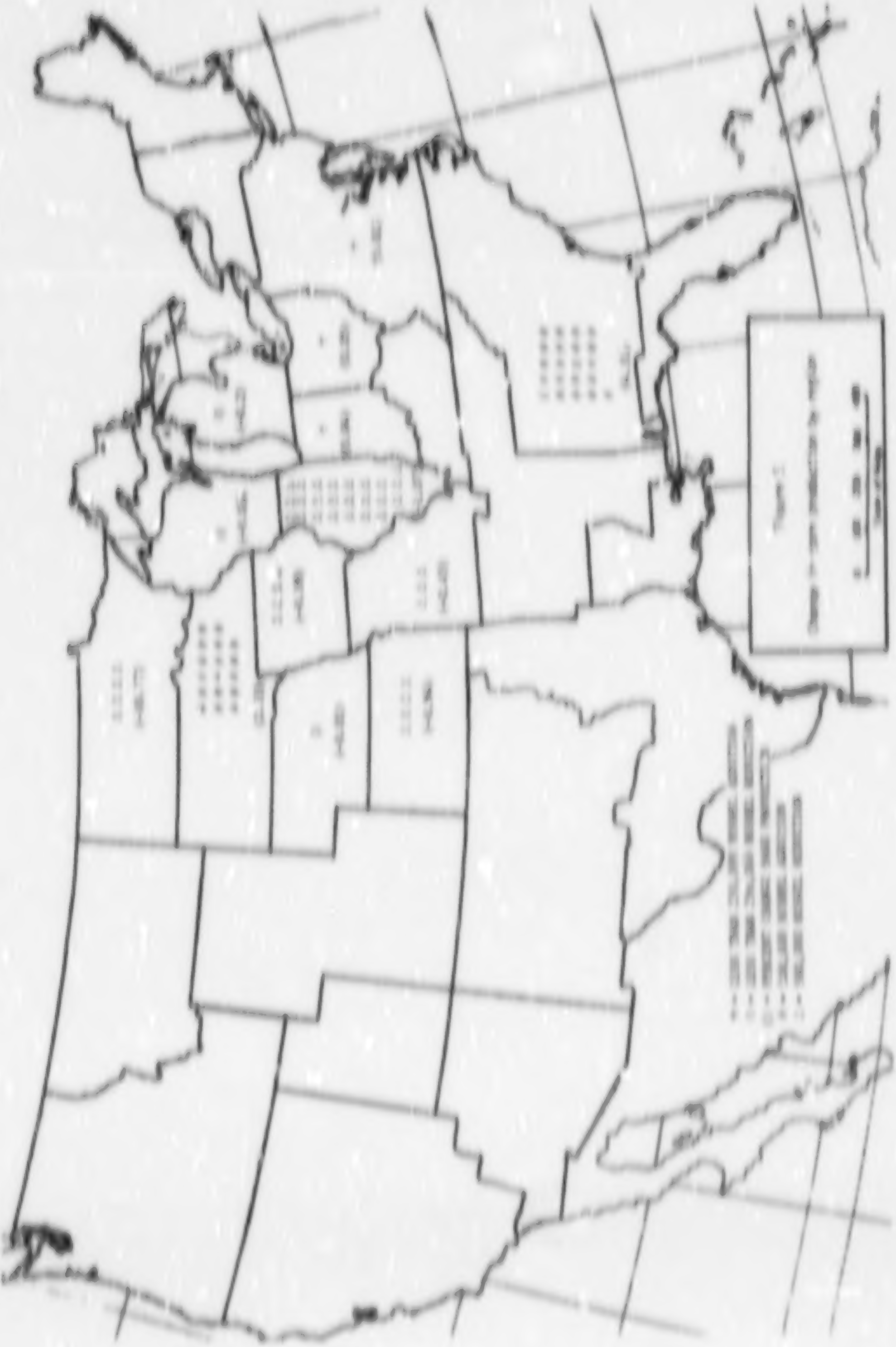
Grain sorghum: No change in any region.

Figure [illegible]
Change in the number of acres of corn by region

0 100 200 300 400
Scale of Miles

TABLE 9. CHANGE IN ACRES OF CORN IN EACH SOIL PRODUCTIVITY GROUP BY REGION

	Group 1	Group 2	Group 3	Total
	(Thousands of Acres)			
Boston				
New York				
Baltimore		0.53		0.53
Atlanta	140.48			140.48
Jacksonville				
Memphis	-9.54		9.54	0.00
Houston				
New Orleans				
Louisville				
Cincinnati	-131.72	80.80	50.92	0.00
Detroit	-33.50	33.50		0.00
Indianapolis				
Chicago	-84.84			-84.84
St. Louis				
Des Moines	-199.72	117.69	82.03	0.00
Milwaukee				
Duluth	-3.28	-28.68	-6.49	-38.45
Minneapolis	-0.76	101.45	-14.78	85.91
Omaha				
Kansas City	-15.35			-15.35
Amarillo				
Billings				
Denver		-0.09	0.09	0.00
Salt Lake City				
Portland				
San Francisco				
Los Angeles				
Total	-338.24	304.41	122.11	88.29



The regional model also permits the analyst to notice some aggregate effects that might otherwise go unnoticed. Reductions in corn production are concentrated in the Western Corn Belt states of Illinois, Iowa, Missouri, Kansas and Nebraska and the Lake States of Michigan, Wisconsin and the northern part of Minnesota. Most of the increases in corn production take place in the Southeast and southern Minnesota with lesser amounts in the eastern Corn Belt and Middle Atlantic states (Figures 4 and 5). If the production shifts were of larger magnitude, these regional aggregations of changes could have significant impacts on the location of feeding and processing.

The minor changes in total production of each crop shown by the use of this model would have no effect on the national price levels. Any regional price changes would be so small that minor changes in interregional grain transportation would offset them. Because there is virtually no change in the farm level prices of grains, there would be no change in consumer level prices that could be attributed to the pesticide decision and thus, no change in consumer surplus in this example. There is, therefore, no economic production effect, per se, to incorporate into a final benefit-cost calculation from this example. The procedure for making such a calculation from farm level price changes is available and may be needed in other examples or after adjustment of the model to take care of weaknesses discussed in the next section.

Weakness of the Model

The model used to illustrate points in this section had weaknesses which were acknowledged by the developers, but which should be eliminated or reduced in future applications. The three major concerns involve (a) the lack of regional demand equations, (b) a need for further refinement in production regions, and (c) a need to refine the "flexibility charges" to more accurately reflect the production adjustments of farmers.

The model used for this illustration assumes that demand for agricultural products is primarily a function of population with adjustments for price and income. The regional demands, therefore, are fixed exogenously based on population projections. While the assumption and procedure are basically correct and the errors in using them are inconsequential in the example used, it is possible to have serious difficulty or substantial error in other applications. Since total production of any agricultural crop was not significantly changed in the "typical case" of a restriction on the use of chlorinated hydrocarbon pesticides on corn, there were no significant changes in product prices. Thus, the inability of the model to adjust quantities demanded to reflect changes in price was not harmful. It is possible that a future pesticide decision will cause significant price changes. The analysis in such a case will be in error if the model does not include a demand function with price as a variable. In any case, the analyst does not know ahead of time whether price changes will be significant or not. Continued work on this aspect of the model is needed.

Due to the nature of specifying a single production equation for each producing region, it was not logical to report production results by the 129 production regions, but rather, they were aggregated into the 27 consuming regions. This technique overcame problems associated with data availability and gives the analyst a view of statewide changes for most of the important corn growing region. As was noted above, however, some of the changes in production that are insignificant when viewed relative to the entire region (for example, the 11.9 million bushel reduction in corn production in the Chicago region is only about one percent of total corn production in the region) might be important if they are concentrated in one area of the region. The 179 production regions of the model seem appropriate for most analysis. Efforts should be made to develop a model that gives useful results at that level of geographic disaggregation.

The third weakness of the model concerns the need to refine the "flexibility charges" which were inserted to reflect reluctance of farmers to make major changes in acres devoted to specific crops. This feature is unique to the EPA model and is an appropriate technique to add realism to the model. We agree with the developers of the model, however, that the specific set of flexibility charges used in this example may have been too high. This resulted in less production response to the pesticide decision than one would have expected. While this weakness does not negate the usefulness of the results from this application, it does indicate the need to refine the technique. Either the charges need to be adjusted or the threshold levels of production change at which the charges become effective need to be changed.

In spite of the weaknesses discussed, the model shows how an analyst can examine likely adjustments in production and cropland use in a highly interrelated agricultural system. When a pesticide decision results in significant shifts in production, the modeling method of the opportunity cost approach permits consideration of regional changes which can affect other businesses in the community. This more comprehensive analysis increases the confidence of the public in the adequacy of the review leading to the pesticide-use decision.

SUMMARY AND COMMENT ON RECENT LITERATURE

Two major approaches to evaluating likely economic production shifts have been discussed. One is the alternative cost approach which assumes that there will be no change in the amount produced. The other is the opportunity cost approach which measures changes in consumer surplus resulting from the restriction using market models or budgeting approaches. The two methods differ greatly in the conceptual acceptability and the data requirements. The alternative cost is less satisfactory from a conceptual standpoint since it assumes that the demand for a product is perfectly inelastic. It has the advantage of requiring much less knowledge of production relationships and production data. In general the alternative cost approach is less accurate but less expensive, whereas the opportunity cost approach is more complete and more accurate, but more expensive. For most applications to decisions involving pesticides used on major agricultural crops, the opportunity cost approach should be followed.

The literature dealing with the effects of pesticide use decisions on agricultural crop production is quite large. This is not surprising since 59 percent of all pesticides used in 1971 went into farm production (2) and a decision to terminate registration of a pesticide could have a large at-the-farm impact. In addition, a pesticide's position as an input into the production function of a product which has a market-derived value makes relatively easy the calculation of the costs of restricting the use of pesticide. Various agricultural economists have addressed themselves to the economic effects of a ban on farm uses of heptachlor (13), lindane and BHC (14), chlordane (31), phenoxy herbicides (25), 2, 4, 5-T in rice production (26) and aldrin in corn production (15). All of these reports found that the direct impact on the farmer would be in the areas of higher production costs, as the farmer switched to other methods of insect and weed control, and yield losses where the alternative methods were less effective. Gerlow (26) pointed out the costs of reduced quality due to weed seeds in rice.

There is some question as to how precise the above measurements of increased production costs might be. Casey and Lacewell (9) used a survey method called the "Delphi" technique to obtain estimates of the yield and cost changes in the production of cotton due to pesticide restrictions. Briefly, this technique consists of polling a number of experts as to their opinion on the question, tabulation of their responses around the median response, and return of the questionnaire to each expert for revision and comment. Through an iterative procedure, the range of the responses is reduced and the final concensus is taken as the best measure available. The obvious problem with the Delphi technique is that the final result is not testable for accuracy. However, in the absence of better data with which to estimate the production function and thus the marginal value of the pesticide, the Delphi technique may be quite cost effective.

In addition to measuring the increase in production costs on the existing output, a measure of the effect on consumer prices must be made. This is because the correct method of calculating the benefit of allowing a pesticide to continue in use is to sum the increased costs of production of the existing output if the pesticide use is cancelled and the area of the triangle of consumer surplus (which represents the excess of value over cost of the decreased output demanded at the higher price). Davis *et al.* (12) in addition to making estimates of the increase in production costs, predicts that the effect of the consumer will be small if effective alternative methods of control are available, because for most farm products pesticides are a small part of the total input. Chapman (11) concurs with this view and adds that the percentage increase in consumer costs would be less than the percentage increase in farm costs due to the large segment (about two-thirds) of consumer cost which is accounted for by processing and marketing costs. He also points out that there is a wide variation in the percentage of processing and marketing costs according to the commodity involved; thus the consumer surplus estimations must be made on a per-commodity basis.

Several studies have looked at the effect of increasing the acreage devoted to production of a commodity as a substitute for the use of pesticides. Delvo (15) held the price/bushel of corn to the consumer constant by increasing land use. Pimentel and Shoemaker (39) investigated the effect of a *complete* elimination of chemical insecticides on cotton and corn production and found that when combined with an elimination of government acreage controls the costs of production of these commodities rose 11 percent and 1 percent, respectively and required 2.1 million more acres. This cost increase was considered a maximum because they did not include the benefit from pest control to be derived from biological, cultural, or integrated methods. Their results indicate that acreage controls which restrict the crop from being grown intensively in regions suited to its production affect crop prices more than do restrictions on pesticides. This conclusion is echoed in Dixon *et al.* (16) where it is found that a reduction in pesticide usage can be obtained by movement of crop production to areas where pests are less of a problem; in this case the movement of cotton production from the pesticide intensive Southeast to the Southwest.

BIBLIOGRAPHY FOR ECONOMIC PRODUCTION CHAPTER

1. Andrilenas, P. A. Evaluating the Economic Consequences of Banning or Restricting the Use of Pesticides in Crop Production. In (42), 1970. pp. 48-62.

2. Andrilenas, P. A. Farmers Use of Pesticides in 1971--Quantities. AER-252, Econ. Res. Serv., U.S. Dept. of Agr., 1974.

3. Arnold, Fred T. Mathematical Modelling to Improve the Quality of Economic Impact Assessment: Studies in Pesticide Policy Analysis. Office of Pesticide Programs, U.S. Environmental Protection Agency, Washington, D.C., 1975.

4. Bailey, G. W., R. R. Swank, and H. P. Nicholson. Predicting Pesticide Run-off from Agricultural Land: A Conceptual Model. Journal of Environmental Quality, 3(2): 95-102, 1974.

5. Brewer, M. F. Agrisystems and Ecoculture, or: Can Economics Internalize Agriculture's Environmental Externalities? American Journal of Agricultural Economics, 53(5): 848-858, Dec. 1971.

6. Carlson, Gerald A. A Decision Theoretic Approach to Crop Disease Prediction and Control. American Journal of Agricultural Economics, 52(216), May 1970.

7. Carlson, Gerald A., and E. N. Castle. Economics of Pest Control. Pest Control: Strategies for the Future. National Academy of Science, Washington, D.C., 1972, pp. 79-99.

8. Casey, J. E. and R. D. Lacewell. Regional and Aggregate Economic Impact of Withdrawing Alternative Pesticides from Cotton Production. Tech. Article 10961, Agr. Exp. Sta., Texas A & M University, College Station, Texas.

9. Casey, J. E. and R. D. Lacewell. Estimated Impact of Withdrawing Specified Pesticides from Cotton Production. So. J. of Agr. Econ., 5(1): 153-159, 1973.

10. Casey, J. E. and R. D. Lacewell. Pesticide I: A General Model to Estimate Regional and Aggregate Effects of Pesticide Withdrawals. Departmental Program and Documentation 73-3, Dept. of Agr. Econ. and Rural Soc., Texas Agricultural Experiment Station, College Station, Texas, September 1973.

11. Chapman, Duane. The Economic Consequences of Regulation or Prohibition of Agricultural Chemicals in American Agriculture. In Environment and Quality of Foods, P. L. White and Dianne Robbis, eds. Futura Publishing Co., Mt. Kisco, N.Y., 1974.

12. Davis, V. W., A. S. Fox, R. P. Jenkins, and P. A. Andrilenas. Economic Consequences of Restricting the Use of Organochlorine Insecticides on Cotton, Corn, Peanuts, and Tobacco. Agricultural Report No. 178, USDA Economic Research Service, 1970.

13. Delvo, H. W., *et al.* Economic Impact of Discontinuing Farm Use Heptachlor. ERS-509, Econ. Res. Serv., U.S. Dept. of Agr., 1974.

14. Delvo, H. W., *et al.* Economic Impact of Discontinuing Farm Use of Lindane and BHC. USDA, ERS #524, 1973.

15. Delvo, H. W. Economic Impact of Discontinuing Aldrin Use in Corn Production. ERD-557, Econ. Res. Serv., U.S. Dept. of Agr., 1974.

16. Dixon, D., P. Dixon and J. Miranowski. Insecticide Requirements in an Efficient Agricultural Sector. Review of Economics and Statistics, 55(4): 423-432, 1973.

17. Edwards, W. F. Economic Externalities in the Agricultural Use of Pesticides and an Evaluation of Alternative Policies. Ph.D. Thesis, University of Florida, 1969.

18. Edwards, W. F. Economic Externalities in the Farm Use of Pesticides and an Evaluation of Alternative Policies. In (42), 1970. pp. 63-70.

19. Eichers, T. P., *et al.* Quantities of Pesticides Used by Farmers in 1966. Agricultural Report No. 179, USDA, Economic Research Service, April 1970.

20. Farris, D. E., and J. M. Sprott. Economic and Policy Implications of Pollution from Agricultural Chemicals. American Journal of Agricultural Economics, 53(4): 66-72, Nov. 1971.

21. Fisher, L. A. The Economics of Pest Control in Canadian Apple Production. Canadian Journal of Agricultural Economics, 18(3): 89-96, Nov. 1970.

22. Fox, A. S., *et al.* Extent of Farm Pesticide Use on Crops in 1966. Agricultural Economics Report No. 147, USDA, Economic Research Service, October 1968.

23. Fox, A. S. Economic Impact of Restricting Herbicide Use. Journal of Environmental Quality, 1(4): 349-352.

24. Fox, A. S. Economic Consequences of Restricting or Banning the Use of Pesticides. In (42), 1970. pp. 34-48.

25. Fox, A. S., *et al.* Restricting the Use of Phenoxy Herbicides--Cost to Farmers. Agricultural Economics Report No. 194, USDA, Economic Research Service, 1972.

26. Gerlow, A. R. Economic Impact of Cancelling the Use of 2, 4, 5-T in Rice Production. USDA, ERS #510, 1973.

27. Giese, R. L., R. M. Peart and R. T. Huber. Pest Management. Science, 187: 1045-1051, 1975.

28. Hall, Darwin C., and Richard B. Norgaard. On the Timing and Application of Pesticides. American Journal of Agricultural Economics, 55(2): 198-201, May 1973.

29. Headley, J. C. Estimating the Productivity of Agricultural Pesticides. American Journal of Agricultural Economics, 50(1): 13-23, 1968.

30. Headley, J. C., and Lewis, J. N. The Pesticide Problem: An Economic Approach to Public Policy, Johns Hopkins Press, Baltimore, Md., 1967.

31. Jenkins, R. P., *et al.* Economic Impact of Discontinuing Farm Use of Chlordane. AER-231, Econ. Res. Serv., U.S. Dept. of Agr., 1972.

32. Jones, Lawrence A. Insuring Crop and Livestock Losses Caused by Restricted Use: An Appraisal. USDA, E.R.S. #512, January 1973.

33. Langham, M. R. Potential for Applying the Dade County Pesticide Model to a Wider Geographic Area. In (42), 1970. pp. 71-79.

34. Langham, M. R., J. C. Headley and W. F. Edwards. Agricultural Pesticides: Productivity and Externalities. In Environmental Quality Analysis, A. V. Kneese and B. T. Bower, eds. The Johns Hopkins Press, Baltimore, Md., 1972. pp. 181-212.

35. Metcalf, Robert L. Agricultural Chemicals in Relation to Environmental Quality: Insecticides Today and Tomorrow. Journal of Environmental Quality, 1(1): 10-14, Jan. - March, 1972.

36. Mighell, R. L. Economic Research on Pesticides for Policy Decision-making. Econ. Res. Serv., U.S. Dept. of Agr., April 1971.

37. Nagadevara, U. S., E. O. Heady and K. J. Nichol. Implications of Application of Soil Conservancy and Environmental Regulations in Iowa Within a National Framework. CARD Report 57, Center for Agricultural and Rural Development, Iowa State University, Ames, Iowa. 1975.

38. O'Connell, P. F., and H. E. Brown. Use of Production Functions to Evaluate Multiple Use Treatments on Forested Watershed. Water Resources Research, 8(5): 1188-1198, Oct. 1972.

39. Pimentel, David and Christine Shoemaker. An Economic and Land Use Model for Reducing Insecticides on Cotton and Corn. Environmental Entomology, 3(1): 10-20, 1974.

40. Smith, V. L. Economics of Production from Natural Resources. American Economic Review. 58: 409-431, June 1968.

41. U.S. Environmental Protection Agency, Economic Analysis Branch, Criteria and Evaluation Division, Office of Pesticide Programs, Detailed Information on Linear Programming Analysis of Organochlorine Suspension of Corn Use in the United States, September 1975.

42. U.S. Department of Agriculture, E.R.S. Economic Research on Pesticides for Policy Decisionmaking. Proceedings of a Symposium, April 27-29, 1970, Washington, D.C., 1971.

43. Van der Plank, J. E. Disease Resistance in Plants, Academic Press, New York, 1968.

44. von Rumker, R. E. S. [illegible]un and F. Horay. Substitutes for Aldrin, Dieldrin, Chlordane and Heptachlor for Insect Control on Corn and Apples. (in preparation) Office of Pesticide Programs, U.S. Environmental Protection Agency, 1975.

CHAPTER 4

HUMAN HEALTH*

To begin a discussion of the methods of measuring the human health cost of pesticides it must first be recognized that until recently there were no studies which specifically linked a pesticide to any chronic debilitating disease. The only documented human effect had been the accumulation of certain long-lived pesticides in human tissue and body fluids (49). The recent discovery of chronic health effects connected to the production of the pesticide Kepone and the increased use of short-lived, though more toxic organophosphate and carbamate pesticides, however, makes the development of a technique to estimate the cost of early death or chronic illness important. Also, in the event that the suspected link between pesticides and cancer is established, the existence of a human "value of life" technique will make the benefit-cost procedure more flexible.

REVIEW OF LITERATURE

The various measures that have been used to estimate the value of life have been criticized by Mishan (38). Lave and Seskin (27) used the present value of foregone earnings to estimate the cost of dying due to air pollution. This technique only measures the individual's contribution to Gross National Product and gives no weight to the preference of retired or unemployed individuals. Fromm (17) measured an individual's value of life by the amount of insurance taken out to insure against a particular risk of death. This measure is incorrect since failure to insure implies placing a zero value on life. The purchase of insurance does not change the probability of death; it only alters the distribution of assets if death occurs.

The correct technique to measure the value of life is to estimate the aggregate consumer surplus involved in the reduction in the mortality rate. This maintains consistency with other measures of societal welfare. The correct question to ask then is, "What is the compensation required to have the person do without the decreased probability of dying?". One method for answering this question is to observe risk compensation in the market. If agricultural workers are paid extra for handling hazardous pesticides, this could serve as a measure of the compensation necessary to assume the higher risk. Of course, this measure would not be applicable to the general population whose risk preference would be likely to be different.

---

*Principal researcher in this section is Mr. F. Roger Tellefsen, Center for the Study of Environmental Policy, The Pennsylvania State University. This section was prepared under the supervision of Dr. Donald J. Epp, Principal Investigator.

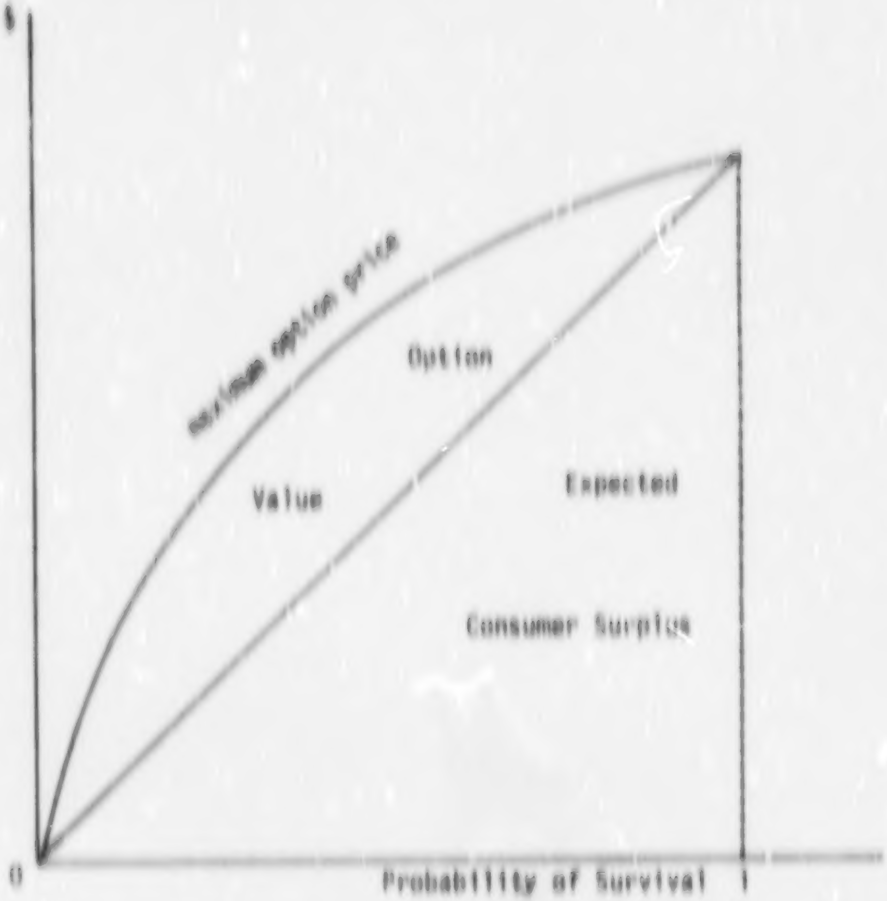
Feldstein (16) points out the distributional problem with this willingness to pay method. Equal weights are assigned to each person's willingness to pay, which may not be true. Feldstein suggests a weighting technique be employed by the decision makers after the benefit-cost study is done.

One human health effect to which it is difficult to impute a cost due to the long time interval over which it acts is that of pesticide induced mutations. While it is true that no human mutations can be traced to pesticides, it is argued (by Epstein and Legator, (11)) that the human genetic material is not so different from that of test animals in which pesticides have caused mutations. Also, they argue that due to high living standards and improved health care, the process of natural selection, which once kept an approximate equilibrium between spontaneously occurring mutant genes and non-viable older mutant genes, is no longer in effective operation. This combined with the near eradication of infectious diseases leads them to the conclusion that "in the future our medical problems will be increasingly of genetic origin." Lederberg (30), using the discredited present value of foregone earnings approach to calculate the cost to society of a one percent increase from the natural mutation rate, comes up with a rough estimate of $200 million/year.

The concept of option demand would seem to be relevant to the discussion of the human health effects of pesticides. In the presence of certainty, consumer surplus will measure the benefits to the consumer. However if there is 1) uncertainty about the future demand for the commodity or service, 2) an element of irreversibility in the decision to supply the commodity or service in the future, and 3) no way for the resource owner to exclude those who do not pay for the benefit, an additional element of benefit must be added to the measure of consumer surplus. Cicchetti and Freeman (2) present a theoretical explanation of option value and show that at low levels of certainty with a large number of users, to not consider option value benefits would "result in significant understatement of benefits." Their model is shown by Figure 6.

THE HUMAN HEALTH MODEL

The human health model follows a logical progression with well defined data needs and several decision points (see Figure 7). We begin with the question of whether or not health problems can be traced to the chemical pesticide being evaluated. The answer can come from either human epidemiological studies or from laboratory animal testing. Care should be taken to design the screening procedure to minimize the possibility of passing a chemical as "safe" because of inappropriate choice of laboratory animal or sample size.



Source: Cicchetti and Freeman (2, p.536).

Figure 6. Option demand.

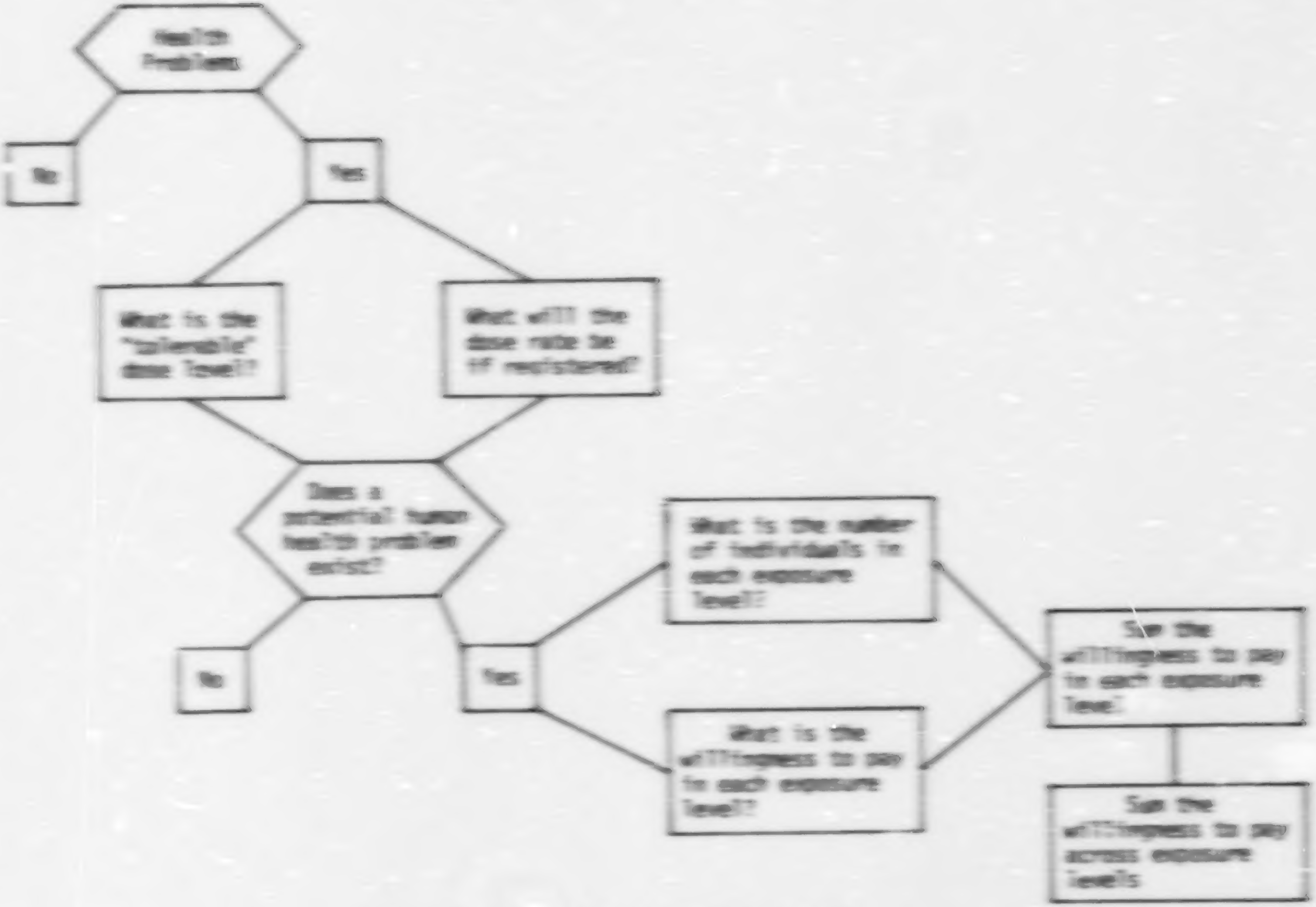

Health Problems
No
Yes
What is the "tolerable" dose level?
What will the dose rate be if registered?
Does a potential human health problem exist?
No
Yes
What is the number of individuals in each exposure level?
What is the willingness to pay in each exposure level?
Sum the willingness to pay in each exposure level
Sum the willingness to pay across exposure levels

Figure 7. Human health assessment model.

If the screening procedure shows the chemical does cause chronic health problems, the assessment of those problems as they affect humans may begin. A decision rule must first be established which delineates a "tolerable" level of risk. Some decision rules may be legislatively mandated--the "Delaney Clause"* for example. In instances where no legislated levels exist, administrative decisions as to what constitutes "virtual safety" will have to be made.

With a tolerable risk level decided upon, the dose level which gives that level of safety can be found. This "tolerable" dose level is compared with the dose rate expected if the pesticide is registered. In the case of a pesticide which has been previously registered, this expected dose rate can be established fairly accurately by examining human monitoring studies conducted by Office of Pesticide Programs of the Environmental Protection Agency (EPA). Additional information can be obtained from the Total Diet Studies conducted regularly by the Food and Drug Administration (FDA). The FDA study samples "market baskets" of food representative of the dietary habits at the sampling location. The nationwide scope of the project permits an accurate estimate of the pesticide residue load through food contamination to which the average American is exposed. However, since pesticide residue can reach the human body from sources other than food (for example, pesticide residues can be transported through the atmosphere as a particle and enter the body through either respiration or absorption through the skin), the Total Dietary Study figures can only be taken as a partial estimate of total body intake. It should be made clear that the desired information is the *total* body intake from all sources.

For those registration decisions which must be made on pesticides which have not been previously used, estimation of the expected dose rate is much more difficult. Knowing the chemical and biological attributes of the pesticide, if an analogue pesticide exists which has previously been registered, it is possible to use that dose rate as a first estimate of the dose rate which might be expected if the registrant pesticide were allowed into use. Taking the application rate of the analogue pesticide and comparing this with the dose rates shown by human survey gives a proportion which can be used in this procedure. Estimates of the quantity of the registrant pesticide (taken from the production model) can be used with the proportion obtained from the analogue pesticide to estimate what the dose rate will be. If the pesticide is registered under these data conditions, special care should be taken during the ensuing five-year registration period to monitor the actual dose levels in the human population. If the levels found to exist during actual use exceed those levels estimated and used in the human health evaluation procedure, a strong case may be made for a reevaluation of the pesticide's human health effects--especially if the initial benefit-cost ratio was marginally in favor of registration.

---

*The Delaney Clause is part of the Food, Drug and Cosmetic Act of 1958 and its implications are described in the section headed "'No Positive Result' Method" (p. 55).

Having established what the "tolerable" dose level is and what the actual dose rate is, it is possible to decide if a potential human health problem exists. If the expected dose rate is above the "tolerable" rate, assessment of the degree of health damage must be performed.

It is clear that it is impossible to speak of a "single" dose exposure level. Keeping the discussion on a conceptual plane, a production worker who deals with the pure active ingredient is likely to be exposed to higher levels of the pesticide than the population at large. Whether the production worker's exposure level is also higher than the pest control operator's or the farm laborer's level is not so clear. However, in all probability, the sub-population should be ranked according to exposure levels. The ordering will depend on the characteristics of the pesticide, the methods and conditions under which the pesticide is manufactured and formulated, and the intensity and geographic breadth with which the pesticide is applied.

We have divided the total population into sub-populations (refer to Table 5) which can logically be expected to have different opportunities for pesticide exposure. It is unlikely that each member of a sub-population will have the same exposure level, however, due to the non-homogeneity of job risk within a sub-population. This makes an estimate of the number of people in each exposure level very difficult to make since, theoretically at least, there can be an infinite number of different exposure levels. As the number of delineated exposure levels increases, the number of individuals in each level decreases (the total population remaining the same). Under the present state of monitoring for human intake of pesticides it is unfortunately impossible to estimate an individual's exposure level precisely, making moot the problem of an infinite number of exposure levels.

The sub-populations should be broad enough to allow a meaningful number of individuals to be included. The best estimates of dose exposure levels from monitoring studies should be used if available and, when combined with the number of individuals in each aggregated grouping, gives the number and characteristics of individuals in each exposure level.

Quantification of the human health effects embodied in the various exposure levels depends on the theoretically correct concept called "willingness to pay." This method is described in the section headed "Willingness to Pay" (p. 66). Briefly, it measures the amount an individual would be willing to pay to achieve a small change in his or her mortality rate. As an *ex ante* concept, it can be answered by the individual either directly or through the amount he is willing to be compensated for undertaking higher risks.

Two individuals who face the same possible change in mortality rate may not agree on the amount they would pay to attain the improved position. A major difference between them might be different attitudes toward risk. It is to control for this factor that the population has been broken down along occupational lines. It is felt that those who engage in the direct production and usage of pesticides are aware of the increased danger and therefore have a different risk function than individuals who avoid occupational pesticide exposure. It is then obvious that if subgroups and exposure levels correspond, the willingness to pay for a given change in mortality rate may not be transferable among subgroups. Ideally then, a complete willingness to pay schedule for each subgroup (risk preference level) should be established.

Once the willingness to pay schedule for a subgroup is established, the dose exposure level is translated into a mortality rate. The average willingness to pay within that subgroup is taken as the amount which would be paid to move from the higher risk level to the "tolerable level of safety" embodied in a lower dosage. Summing this amount across the number of individuals in the subgroup results in the amount that that subgroup would be willing to pay to reach the "tolerable level of safety." Performing the above procedure for each subgroup and summing across all subgroups results in an aggregate willingness to pay figure for the expected lower mortality rate.

It should be pointed out that for those individuals who are exposed to pesticides by their own choice (predominantly those involved in pesticide production, formulation, and application), only that portion of their willingness to pay which is not returned by their compensation (earnings, employer paid health and life insurance, retirement benefits, etc.) should be included in the benefit-cost procedure. To do otherwise would be to double these items since they are all costs to the employer and enter into his cost of production. At the present, it is assumed that these employees perceive the riskiness of their occupations and that they receive compensation for their perceived risk. Whether they are successful of course depends on their market power. Lacking data as to the extent of their success, the assumption that their willingness to pay has been completely returned to them can be made.

This effectively reduces the population to be considered in the benefit-cost analysis to those whose injury or residue intake is due to non-occupational exposure. Their willingness to pay is not returned to them in their earnings, and they have little choice as to their exposure level.

MEASURING PESTICIDE RELATED ILLNESS

Establishing a scientific basis for evaluating the hazards of chemical pesticides on human health is a major problem. If a causal connection between a pesticide and human health can be demonstrated, the estimation of the benefits or costs of that effect can proceed according to the method to be described. Establishing a connection through actual observation of humans exposed to pesticides is the job of epidemiologists while biostatisticians attempt to establish a connection via animal testing.

Acute Injury

Acute injury due to pesticide poisoning does occur, causing disruption in the life of the victim. Pain, nausea, fever, and death are some of the results of pesticide poisoning. The fact that virtually all poisonings are the result of accident or human error (the suicidal fringe of the population which uses pesticides to end a life is disregarded here) does not lessen the need to consider this aspect of pesticide use. To suppose it does invites an analogy to automobile injuries. Most injuries in automobile accidents are the result of human error. Yet one does not consider these injuries of little consequence. Acute pesticide injuries may be several orders of magnitude fewer in number, but their individual effect can be just as great.

Assessing the risk of pesticide poisoning is possible by referring to published studies. The EPA is currently conducting a National Hospital Survey which will greatly improve data in this area. The difficulty, of course, is in apportioning the risk among various groups. For example, the data may indicate that the nationwide risk of pesticide poisoning is 7 in 100,000. However, the risk faced by the agricultural worker in vegetable fields may be hundreds of times greater.

In order that the risks of acute pesticide poisoning may be estimated, the method of exposure as well as the specific chemical involved must be determined. Once these parameters have been established, it is possible to identify groups which are at risk and to rank these groups according to their probability of acute poisoning.

Chronic Injury

Health problems making their appearance in man only after a long period pose different problems. First there is the problem of recognition. In test animal studies, a subject can be sacrificed at regular intervals and an autopsy performed in order to determine if a disease is developing. For obvious reasons this course of action is neither feasible or desirable for human studies. Furthermore, the short life span and rapid reproduction of most species of test animals allows any effects to be observed over several generations in a fraction of the time elapsed for a comparable human test. Thus, any detection system for chronic effects in humans must await the development of the disease to the acute stage. At this advanced stage it is often too late to cure the victim of disease. Another problem is that the information gained from one case is limited to the recognition that one person has contracted a disease which has developed over some time period and whose cause is uncertain. This brings up the problem of attribution.

It is often very difficult to attribute a chronic health effect in humans to the correct cause. This is due to the lack of experimental controls. In the case of an ill person, the problem could be ascribed to genetic, occupational, environmental, dietetic or some other factor to which the individual has been exposed. Until a body of evidence builds up linking the illness to one or more of these factors, epidemiologists are unable to say with any degree of assurance what the causal connection might be.

Animal testing programs have the advantage of allowing controls to be placed on the genetic, environmental, and dietetic factors, thereby holding to a minimum the extraneous effects which so often confuse human investigations. Typically they are performed at higher dose levels than are encountered by humans in the environment. This allows a chemical to be screened for harmful effects more quickly by using the experimental direct correlation between dose level and chronic effects. Thus, experimental animals exposed to high but sublethal doses of a suspected oncogen have a greater probability of contracting a disease above the spontaneous rate within the time frame of the experiment and establishing the level of danger of the chemical. The high dose levels are also necessary in order to keep the screening procedure within financial bounds and still maintain statistical precision. This follows from the well known proposition that information is not costless.

In the case of mammalian studies the cost of increasing the number of experimental subjects climbs rapidly due to the expense of housing, feeding, supervising, and final analysis of larger numbers of animals. Furthermore, experiments large enough to detect high incidences of harmful effects at low dose rates have an increased chance that faulty laboratory technique may be introduced merely by virtue of the difficulty in managing an experiment of such size. These factors mitigate against the use of huge numbers of test animals for screening purposes, and, when combined with the earlier mentioned association between dose and effect, imply that the most information for resources expended is obtained in high dose exposure testing.

TECHNIQUES FOR EVALUATING RISK LEVELS

Given this experimental state of affairs, what should be the technique for establishing human risk? It is obvious that dose levels which induce mutations, cancer, tissue growths or any other harmful effect in test animals cannot be allowed in the human experience. But what of the lower dose levels to which humans may be exposed which have not been directly tested on laboratory animals? What methods exist to evaluate the human risk at these low doses? The next section considers the possible approaches.

"No Positive Result" Method

One approach is to remove from the human environment any chemical which has been shown to cause major harmful effects in humans or animals. This is the approach taken against cancer causing food additives by the so-called "Delaney Anti-Cancer Clause" of the Food, Drug and Cosmetic Act of 1958. It prohibits the addition of any additive which has been shown to induce cancer in man or animal. It further prohibits the addition of any cancer inducing additive to food-producing animals unless there are zero residues of the additive present when the animal is consumed as human food.

Such a draconian decision rule would be satisfactory if one could conduct the laboratory screening with great statistical precision. With limitations on experimental size, however, the possibility is great that an actual carcinogen could pass this test due to small sample size.

For example, a test result of zero tumors in 100 animals gives a tumor rate of 0 percent. Such a test result would fulfill the requirements of the Delaney clause. However, statistically it can be shown that such a result is consistent with a true risk of 4.5% when an assurance level of 99% is employed.

$$(1-p)^{100} = .01$$

$$100 \log (1-p) = \log .01 = -2$$

$$\log (1-p) = -.02 = 9.98-10$$

$$1-p = .955$$

$$p = .045$$

Increasing the test to 0 tumors in 1,000 animals would only lower the upper limit on tumor risk to .45% (36, p. 460).

In an effort to make up for this deficiency, a fraction of the highest "no-effect dose" is taken as safe for human consumption. This fraction, usually 1/100, has no basis in biology and is merely an arbitrary safety factor. Furthermore, since the "no-effect dose" is an independent variable in the experiment, it opens the possibility that unduly high levels of residues may be allowed because of uninformative experimentation, i.e., the use of low dosages or small sample sizes in the screening experiment. On the subject of "no-effect dose" Weisburger and Weisburger have written:

> "It seems to us a 'no-effect dose' for a carcinogen is a highly relative level which applies only for the precise experimental conditions generated. While similar considerations hold for drugs, the risk is not nearly so intense. More often than not, an improper dose rate for rapidly acting drugs is detected almost immediately and appropriate remedial action can be taken. With chemical carcinogens and their long latent period, the disease condition resulting from inappropriate selection of dose levels and alteration of environmental conditions leading to potentiation may become visible only years after the exposure. At that time remedial action is obsolete and often worthless."*

Our conclusion is that the "no positive result" method with its arbitrary "safety factor" is inappropriate. Its use leads to a situation where the pesticide is banned before any consideration can be made of its true risk. Indeed, risk is never really considered under this method. It is assumed to be reduced virtually to zero by the use of a safety factor. Finally, as previously described, this method tends to reward uninformative experiments by establishing potentially high permitted dose levels.

Mathematical Extrapolation

By mathematical extrapolation we mean any statistical investigation of dose-response relationships beyond the range of relationships which are experimentally observable.

There has been and there continues to be a great deal of discussion among toxicologists and biostatisticians on the subject of mathematical extrapolation models. It appears clearly ill-advised to join the fray over which

---

*Weisburger and Weisburger, *cited* in (5, p. 161).

model is the "best." Each model has its weaknesses as well as its strong
points. The intention instead is to briefly describe the various models and
mention the attributes which lend themselves to the problem at hand.
Finally, a determination is made of a recommended model for this project
based on currently available information. Since the field is changing very
quickly with refinements and new techniques appearing annually, different
models may be used in future studies.

The three extrapolation procedures examined differ mainly in their
prediction of the shape of the lower end of the dose-response curve. The
probit model postulates that the individual members of the population have
differing levels of susceptibility to a carcinogen and that this variability
can be statistically approximated by the normal distribution (the so-called
"bell-shaped curve"). It further postualtes that since the dose-response
curves at low doses are likely to be concave upward to the observed region,
placing a straight line extrapolation with a low slope from the observed
region sets a conservative upper limit to the risk. The *logistic model*
postulates that response is tied to the number of available chemical bonding
sites. As the sites become occupied, the response curve levels off, taking
on a shape best described by a log curve. The *one-hit model* postulates that
a response is induced by a single exposure to a cell by some disruptive agent
such as a chemical. This implies a dose reduction proportional to the risk,
resulting in a linear extrapolation curve from the observed region extending
to zero dose. For example, the dose which resulted in one tumor in 100
animals would be reduced by a factor of 1,000,000 to obtain an extrapolated
upper limit to risk of 1/100,000,000.

The degree of conservatism between these models varies widely. To
illustrate this fact a table from the FDA Advisory Committee on Protocols for
Safety Evaluation (14, p. 433) is cited (in Table 10). It can be seen that
at high levels of estimated risk the models do not differ dramatically in
their permitted dose levels. However, as the estimated risk falls, the
permitted dose levels between models become quite different.

In the absence of experimental results in the low dosage range to
demonstrate the true dose-response curve and thus the true dose-risk relation-
ship, a case can be made for the choice of the more conservative extrapol-
ation models--the one-hit model or the logistic model. If more than one
pesticide contributes to the total risk, in that their actions were similar
to each other, then use of the one-hit rule will minimize the possibility
that their combined "safe" doses will become unsafe. Further, if synergistic
effects between these pesticides possibly exist, the one-hit rule would give
greater protection than a less conservative rule merely by virtue of setting
lower individual safe doses. The true risk of synergistic effects would
have to be investigated via multiple-agent experiments (32, p. 1382).

If biological considerations such as those mentioned above do not exist,
the unnecessary use of very conservative models could lead to regulatory
problems. If the "safe" dosages are almost always so low as to virtually
prohibit any usage at all, the regulatory agency involved, in this case the
EPA, would be under pressure by registrants to devise an alternative pro-
cedure which would be more amenable to the "real world situation." What is

desired is an extrapolation method whose conservatism has basis in biology. This attitude makes defense of any decision based on extrapolation tenable.

TABLE 10. EXTRAPOLATED VALUES OF "SAFE" DOSES FOR THREE DIFFERENT DOSE-RESPONSE CURVES DESCRIBING OBSERVED RESPONSES IN 2% TO 50% RESPONSE RANGE EQUALLY WELL

Estimated risk	Probit	Logistic	One-hit
1/100	4.0×10^{-2}	2.2×10^{-2}	1.4×10^{-2}
1/1000	1.5×10^{-2}	3.1×10^{-3}	1.4×10^{-3}
1/1,000,000	1.4×10^{-3}	9.8×10^{-6}	1.4×10^{-6}
1/100,000,000	4.1×10^{-4}	1.6×10^{-7}	1.4×10^{-8}

SOURCE: F.D.A. Advisory Committee on Protocols for Safety Evaluation (14, p. 433).

The probit model, often called the "Mantel-Bryan model" after the authors of the article (36) which first discussed its use in extrapolation, has several features which indicate its suitability for use in this project. First, it is a linear extrapolation methud which weighs dose against probability making it relatively simple to associate a specific dose with a risk level. To illustrate: Figure 8 shows the estimation of the "safe" dose of a potent carcinogen methylcholanthrene. The solid line is fitted to the data and for the region of observation has a slope of 2.64 normal deviates per log dose. The extrapolation is performed by extending a line with conservative slope of one normal deviate per log dose from the lowest upper confidence limit, in this case that associated with the second dose level. Although the procedure is intended to set maximum permitted dose levels given a prespecified risk level (in the illustration, a risk of 1/100,000,000), there is no reason why the procedure cannot be reverted and used to estimate the risk associated with a prespecified dose level. In this manner the upper limit to risk of exposure to 'x' parts per million (ppm) of pesticide could be estimated from laboratory experiments at much higher dosages.

It should be noted that there are two variables in the Mantel-Bryan method which can change the dose-risk relationship. The most obvious is the slope of the extrapolation line. Mantel-Bryan recommend the slope of one probit per log dose as being conservative from the standpoint that dose-response curves in the observable region are virtually always of steeper slope. A conservative extrapolation line is necessary to guard against the possibility of setting permissable dose levels which are unduly high. Other researchers have argued for steeper extrapolation slopes. The effect of using different probit slopes upon the permitted dose with a given estimate risk of 1/100,000,000 is shown in Table 11. It can be seen that the steeper

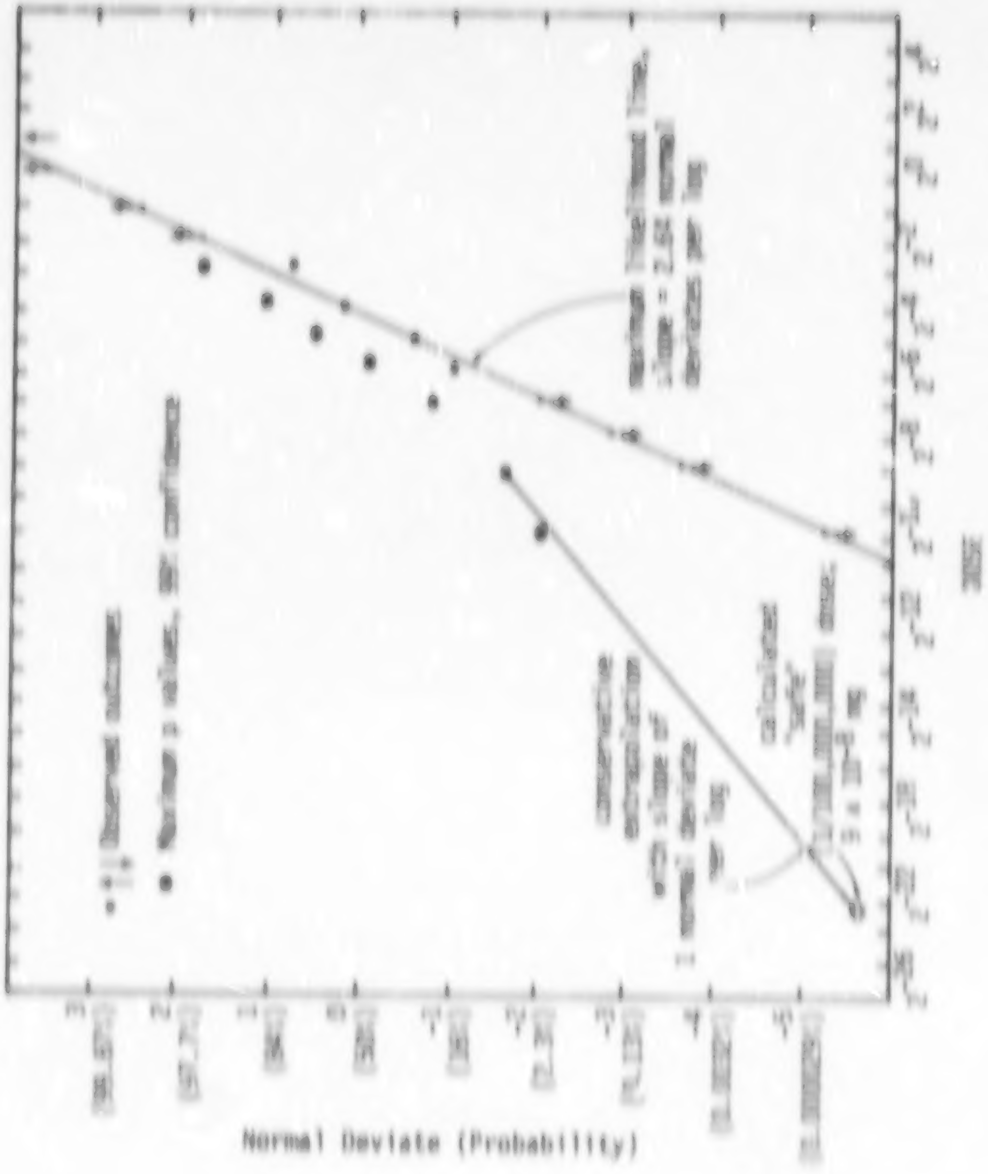


Figure 8. Dose-risk extrapolation using Mantel-Bryan procedure.

slopes allow much higher dose levels--consequently, for a given dose exposure, the use of higher slopes will result in a lower estimate of risk being attached to the dose. A conservative bias on the subject of potential gross health effects should prevail, and in the absence of contrary evidence the unitary probit slope is used to make risk estimates.

TABLE 11. FRACTION OF EXPERIMENTAL DOSE USING PROBIT EXTRAPOLATION WITH DIFFERENT SLOPES FOR AN ESTIMATED RISK OF 1/100,000,000

Observed tumors	Slope = 1	Slope = 1.5	Slope = 2
0/50	1/18,000	1/690	1/135
0/100	1/8,300	1/410	1/91
0/500	1/1,800	1/150	1/42
0/1000	1/1000	1/100	1/32

SOURCE: Crammer (5, p. 164).

The other variable which can effect the dose-risk relationship is the choice of the upper confidence limit. Willingness on the part of the researcher to accept as true a false hypothesis is reflected in the choice of confidence limit. In the case under consideration this translates into a willingness to accept a no-response result as true when that result could have been caused by statistical variability. The choice of a low confidence limit results in a wide confidence interval. Under the Mantel-Bryan model such a choice leads to lower risks being attached to given dose levels than would be the case for a high confidence limit. This feature is illustrated in Figure 9. Mantel-Bryan suggest a 99 percent confidence limit as appropriate for large populations.

A further attribute of Mantel-Bryan is that the method "rewards" good experimentation. Registrants who test their chemicals with large numbers of animals and/or at high dose rates are "rewarded" with high "safe" levels. The increased information which results allows the confidence intervals to become smaller, thereby moving the extrapolation line to the right. Conversely, poor experimentation results in wide confidence intervals and the establishment of low "safe" levels. This is helpful from a registration standpoint since it makes the registrant responsible for setting his own permissable residue levels. From the point of view of the benefit-cost procedure it allows the estimation of risk levels to be made with greater confidence.

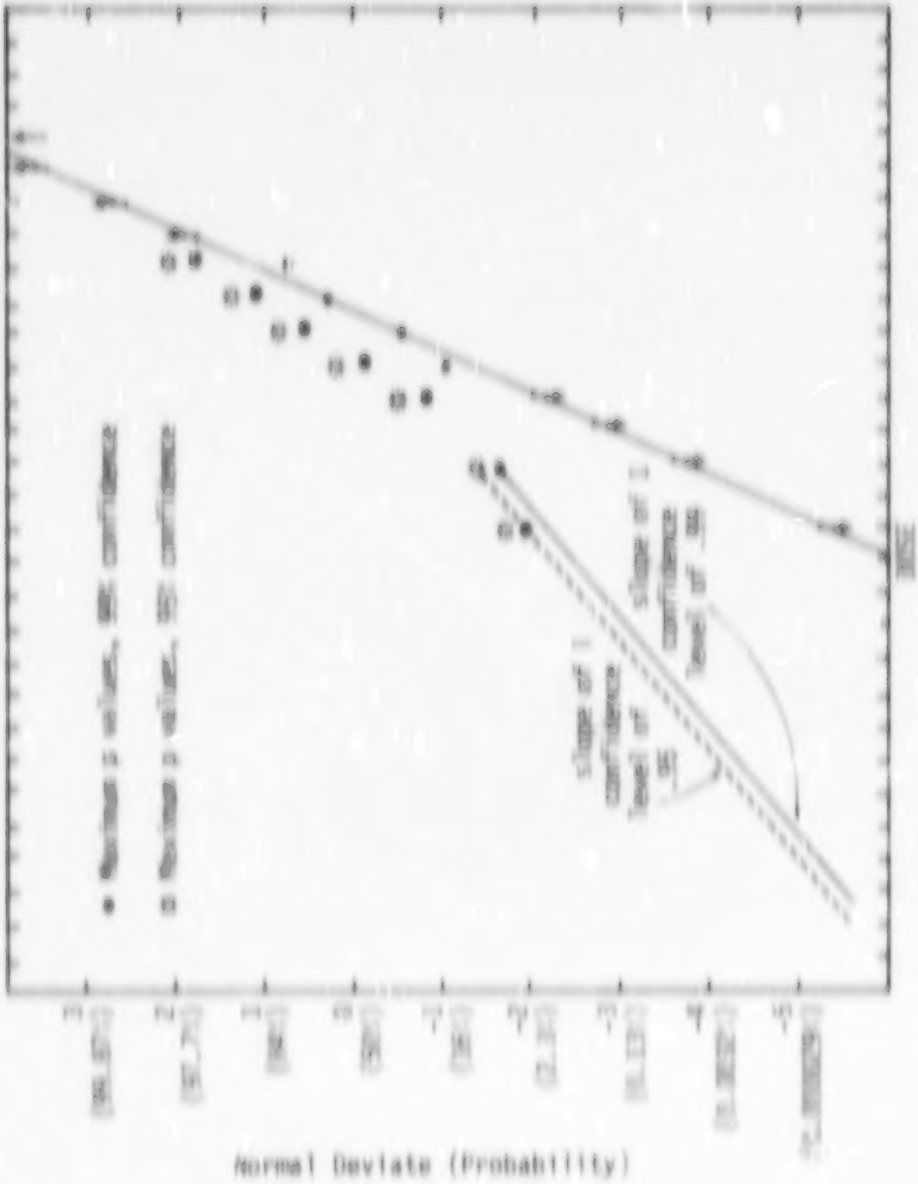


Figure 9. Effect of changing confidence levels in Mantel-Bryan procedure.

Since their 1961 article was published (36), work has been done on the Mantel-Bryan procedure to correct several statistical deficiencies. A recent report (33) details the improvements made. One of the most interesting is the ability to handle several sets of independent data. This allows the risk level associated with a dose to be updated as new experiments are performed. Information from registrants, independent laboratories, and government laboratories can be combined--the result being improved confidence in the extrapolation.

A caveat must be made at this point. Strictly speaking, what is being done by any of the mentioned extrapolatory models is the setting of a "safe" dose for the animal upon which the experiment is performed. For example, if it is a mouse experiment, thus we have set the safe dose to protect mice from a cancer-causing chemical; the problem of translating animal studies into human terms is still present. Unfortunately, mice are not men, and studies which compare the respective metabolic processes of laboratory animals and humans are needed. About all that can be done at this time is to maintain comparability between pathways of exposure in the laboratory animal and man. Various conversion factors have been suggested depending on the way dosages are stated.* The existence of this problem accentuates the need for a conservative extrapolation method in order to protect what must, in the absence of contrary evidence, be assumed to be the most vulnerable species--*Homo sapiens*.

TECHNIQUES FOR EVALUATING A CHANGE IN MORTALITY RATE

The dose-risk extrapolation for chronic effects outlined in the preceding section can be used by the decision maker in a number of ways. Most simply, it can be translated into mortality and morbidity figures. Taking the extrapolated risk and multiplying by the size of the population exposed will give an upper-limit estimate to the number of acute or chronic illnesses which will be expected to result from the pesticide under review.

Constructing an example for illustration, suppose one million farmers were exposed to a pesticide residue of .01 mg and that extrapolation from animal testing associated a risk of 7.5/100,000 of contracting cancer with such a dosage. It then follows that as many as 75 farmers can be expected to contract cancer as a result of their pesticide exposure. It must be emphatically pointed out that 75 is the upper limit to the number of cases which are expected in this example; that is, since the extrapolation procedure aims toward conservative estimates, the true risk (which cannot be known) may be much less than 7.5/100,000.

---

*The surface area rule, for example, relates surface area of an organism to its weight; the surface area is proportionate to 2/3's root of its weight. "If mouse dose is stated in flat amounts, obtain dose for man by multiplying the mouse level by 2/3's root of the weight ratio, i.e., 200." Mantel and Schneiderman, (37, p. 1386f).

If desired, the estimation of the human health effects of a pesticide can be left in these terms. Even if further refinement is desired, this first step serves as a valuable screening stage. Those who are at risk are identified and an estimate is made of the number of potential victims. If the number is small (the determination of "small" to be arrived at by scientific and political concensus), the examination of human health effects can be terminated here. If the number is large, the remainder of the assessment procedure should be completed.

A benefit-cost procedure is most complete when as many of the relevant entities as possible are reduced to a common denominator. This is a difficult objective in areas which are not normally exposed to the equilibrating forces of the market place--one such area being the desired level of human health. It is felt that techniques exist which allow the human health effects to be made commensurable with the output of the production effects section (Chapter 3).

Attempts at systematic evaluation of the value of human life have been made since at least 1931 when an effort was made to measure the costs of World War I to the American people (3). In 1946, Louis Dublin and Alfred Lotka published a work which was designed to find the amount of insurance which an individual should carry ([illegible]). Problems arose when economists began to use variations of Dublin and Lotka's technique to estimate the costs of disease and thus the cost of death due to the disease. Most of these endeavors can be discussed under the heading of "foregone earnings."

Foregone Earnings Approach

Generally, the foregone earnings approach posits that the discounted value of the future stream of earnings is the appropriate estimate for the cost of an untimely death; and, since the analysis is symmetrical, the appropriate estimate for the value of a life *saved* from an untimely death. An advocate of this technique stated that "if an individual had not died or become ill this year, he would have continued to be productive for a number of years. It is the present value of these future losses that is required as an appropriate evaluation measure of the costs of a disease for program planning purposes" (44, p. 438). Immediately, the question arises over what is the correct measure of earnings to be used in the calculation--gross or net.

Those who support the use of *gross* earnings cite the fact that when an individual ceases to receive earnings due to death, production is lost to society. If what is of interest to society is lost production, and the individual decedent's marginal productivity is measured by his or her earnings stream, the loss to society can be expressed as

$$L_1 = \sum_{t=p}^{\infty} Y_t \, P_p^t \, (1+d)^{-(t-p)} \qquad (1)$$

where Y_t is the expected gross earnings of the person during the t th year, P_p^t is the probability in the current (p th) year of the person being alive during the t th year (4, p. 688). Among those who have used the gross estimate of earnings are: Rashi Fein, who estimated the loss to society caused by admissions to mental institutions (12), Herbert Klarman who estimated the loss to society due to syphilis (24), and Dorothy Rice who estimated the loss to society due to illness of all kinds (45).

Those who prefer the use of net earnings disagree with the above approach by arguing that although society loses the production of the decedent, it also does not have to supply goods for the decedent's consumption. Death releases resources to the rest of society for their use. Thus, the appropriate loss to society under this method is

$$L_2 = \sum_{t=p} P_p^t \, (Y_t - C_t)(1+d)^{-(t-p)} \qquad (2)$$

where C_t is the consumption of the individual during the t th period that is expected at time p (4, p. 688). It is easy to follow the logic of this argument. However, closer examination of the equation reveals that it is possible that the cost to society of a death could be negative (i.e. a "benefit") if the individual consumes more than he produces. Retired and unemployed individuals are likely to have such negative life valuations.

Weisbrod points out that the problem here is one of defining the "society" which is to be included in the evaluation. If the definition of "society" excludes the individual whose value of life is being estimated, then the net earnings correctly estimates the contribution he makes to society. If "society" includes the individual, then the gross earnings correctly estimates the contribution he makes to his society. Including the individual whose value of life is being measured in "society" removes the possibility of negative valuations but does admit the possibility of a zero valuation (53, pp. 35-36).

The possibility that the procedure may result in a negative or a zero valuation of life has always been a major criticism of the foregone earnings approach. Individuals whose marginal product is not measured in monetary terms (the largest single example is output from housewives being excluded from Gross National Product measurement), who are temporarily non-productive (those who are unemployed), who are retired from the work force are all victims of a methodological problem which equates their value to society to zero. In the case of housewives, imputation of value to their services at the average earnings of a domestic worker has been carried out (45, p. 429). This imputation results in a lower valuation to causes of illness and death which predominantly affect women, due of course to the low wage of domestic workers relative to the average wage of male workers.

It becomes clear from this brief discussion of the foregone earnings approach that the entity missing from the analysis is the individual at risk.

The procedure dwells on only one portion of an individual's contribution to society (and typically that contribution is measured against the standard of Net National Product). If the goal of society is the maximization of NNP, then assessing the costs or benefits of a health impacting project according to the foregone earnings approach can be rationalized. If society holds other goals as well, then the use of foregone earnings is merely an expedient approximation.

Other Approaches

Another method of estimating the value of life may be thought of as the "insurance premium" approach. In this method the premium paid for insurance to cover a probability of death from a particular cause is translated into the value placed on life. Gary Fromm (17) applied this concept to the case of a traveler who buys flight insurance. If the risk (r) and the corresponding premium (p) are known, then the value of his life is set at $\frac{p}{r}$. A one-tenth of a percent risk with a premium of $10 implies a value of life of $10,000. It has been pointed out that the flaw in this procedure is that the payment of an insurance premium does not alter the probability of death, it only alters the distribution of assets within society if death occurs. Thus, the purchase of insurance can be interpreted only as an indication of the purchaser's concern for his beneficiaries and not as an indication of an individual's value of his own life (41).

Another concept which is often discussed when the value of life is considered is that of quantifying intangibles. Pain, suffering, anxiety and grief are some of the psychic costs which are widely believed to be part of disease and death and should somehow be included when measuring the value of life. Rice takes a common position when she says:

> Intangible or psychic costs of disease, such as pain and grief, are omitted [from her study]. These costs do not directly involve a loss of output and are not readily measurable. Several economists feel that ignoring the intangibles may distort the over-all economic and social costs because the implicit assumption is that the economic value of intangible losses is zero.

It is of course not true that the economic value of intangible losses is zero. These negative factors are an important part of the reason why people have such a high desire to avoid illness and death. The reactions of those individuals who are completely covered by health and income insurance indicate that there is more involved to these individuals than loss of income or high health care costs when death or illness threaten. These personal feelings of the value of avoiding intangible costs should somehow be included.

Inadequacy of Evaluation Techniques

The most telling argument against the previously discussed evaluation techniques is that they are conceptually incorrect for use in a benefit-cost analysis. For whatever reason, they have long been giving answers to the wrong question. It is for this reason that this study is aimed at correcting conceptual inconsistencies and at asking the correct question.

Dan Usher (50, p. 195) discusses three related but distinct questions which can be used to point out the erroneous use of past measurement techniques. Question one, the *insurance question*, asks "What is my life worth to my wife and children?" Question two, the *birth control question*, asks "How much better or worse off would society be if I ceased to exist or had never been born?" Question three, the *valuation-of-life question*, asks "How much would I pay to avoid a small probability of my death?"

The present value of foregone earnings approach answers the insurance question. The amount of money which satisfied L_1 is the amount which will keep his family in the same financial condition as it would have been had he not died. It is the appropriate amount of insurance for him to carry. It says nothing about what his answer to the valuation-of-life question might be since it does not address itself to that concept. The fact that a man has insured himself for the full amount L_1 does not imply that he is now indifferent to death. The same criticism applies to the "insurance premium" approach.

The discounted net earnings approach answers the birth control question. Since L_2 excludes the individual's preferences, as previously discussed, it is quite possible that society would be better off if the individual were to die. Once again, the procedure does not address itself to the valuation-of-life question.

The question that should be asked is the valuation-of-life question--How much will someone pay to avoid a small probability of death? This concept is the theoretically correct one to use in a benefit-cost study since it maintains consistency with other measures of societal welfare; in the case of measuring the value of life, it estimates the aggregate consumer surplus involved in a reduction in the mortality rates. The next section develops this concept in detail.

Willingness to Pay

Discussion of what has come to be known as the "willingness to pay" concept is burgeoning in the recent economic literature. Perhaps the best known advocate is the British economist E. J. Mishan who has written widely on welfare economics (42) and benefit-cost analysis (38). His 1971 article forms much of the theoretical background upon which the breakaway from "foregone earnings" is founded. One of the earlier statements of the willingness to pay concept was made by Thomas Schelling (46). An attempt to apply willingness to pay for auto safety features was made by Lester Lave and Warren Weber (28). They found that auto buyers were, for the most part, rational in refusing to buy certain safety devices. In a major theoretical work (with practical illustration of the effects of imputing value to declining mortality rates) Dan Usher (50) developed a model which estimates $\frac{\partial \text{ Utility}}{\partial \text{ Mortality Rate}}$ subject to the parameters d (rate of discount) and β (the elasticity of annual utility with respect to consumption).

Bryan Conley (4) developed a theoretical model which serves to connect the discounted earnings technique with the willingness to pay technique (at least in terms of relative magnitude). He found that the value of saving a life was greater than lifetime discounted earnings.

The practical difficulty with the willingness to pay concept has been the measurement of the amount individuals are willing to pay for small reductions in the mortality rate. This is unlike the type of purchase decision commonly faced by the average consumer. The concept, however, is not totally alien to consumers. For example, when one purchases, installs, and uses safety equipment, one is changing the probability of death and injury that must be faced. A home fire alarm system lowers the probability of asphyxiation while asleep. The use of automobile seat belts reduces the risk of death or serious injury in an accident. The purchase of safe laboratory approved home appliances gives some assurance against electrical shock. Even the installation of non-slip material in the shower stall affords an improved probability of survival. Other examples where safety is a component of the purchase decision can be found.

Another technique to ferret out people's willingness to pay is to look at the extent to which they are willing to be compensated to incur *higher* risks. This behavior is prevalent in those occupations which have been shown to be actually riskier than the average occupation. In these occupations, a risk premium may be paid to encourage workers to engage in higher risk activities. For example, coal mining is inherently riskier than office work. Within the coal mining industry, underground work is riskier than aboveground work. Underground miners receive extra compensation for working under riskier conditions and this amount can be used to estimate their willingness to pay to obtain small reductions in mortality.

The theoretical equivalency between "willingness to pay" and "willingness to be compensated" can be demonstrated graphically. In Figure 10 we have portrayed a small part of an individual's indifference map. Each point along an indifference curve (U_1 and U_2) yields the same level of utility to the individual. That is, the curves trace the combinations which result in equal levels of satisfaction to the individual. The vertical axis is denominated in money terms (Y) and the horizontal axis is denominated in terms of "quantity of safety" (S). The trade-off is thus income vs. safety and the individual chooses his position subject to the constraints of total income and the "price" of safety.

The following two sections examine the effect of a change in the price of safety. First we examine the case where the price of safety falls. This would be the situation if, using pesticides regulation as the instigating event, the government were to unilaterally ban a pesticide which was linked to increases in human mortality. The individual would then have to pay fewer dollars to obtain the same level of safety.

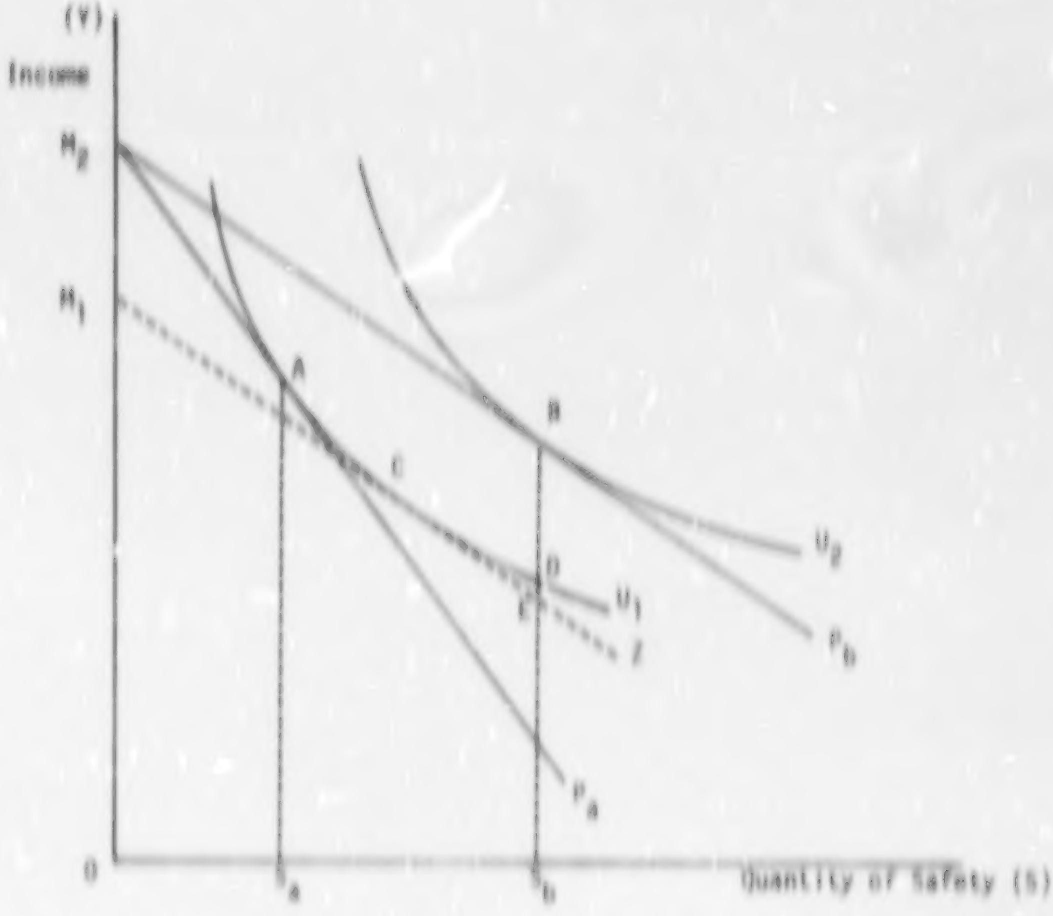

(Y)
Income
M_2
M_1
A
B
C
D
E
U_2
U_1
Z
P_b
P_a
0
S_a
S_b
Quantity of Safety (S)

Figure 10. Safety indifference map.

Decreases in Mortality--

The individual has an income of OM_2 and is initially faced with the price ratio ($\frac{dY}{dS}$) of P_a. He maximizes his utility such that the price of safety (P_a) equals the marginal rate of substitution (MRS) along utility curve U_1 (at point A) and obtains OS_a of safety. With the fall in price to P_b he again maximizes his utility such that P_b equals the MRS along utility curve U_2 (at point B) and obtains OS_b of safety. In the process he has become better off, moving to a higher indifference curve and increasing his consumption of safety. To find out in monetary terms how much better off he is at his new equilibrium point, we construct a line M_1Z with slope of P_b tangent to the lower indifference curve U_1. If the individual's income had fallen by the amount M_2M_1 at the same time the price fell from P_a to P_b, he would have been just as well off at C as he was at A (since he finds himself on his original indifference curve U_1). Since income remains constant and the price falls to P_b, then M_2M_1 measures how much better off B is than A. Hicks (19) called this amount the "compensating variation" for the price fall.

From this point it is possible to describe the change in the consumer surplus which was obtained in the move from quantity OS_a to OS_b. The new lower price allows the individual to obtain OS_b of safety. The net benefit to the individual of the higher quantity is given by the vertical distance between indifference curve U_1 and indifference curve U_2 at the new quantity obtained (OS_b). This net benefit (or change in consumer surplus) is represented by BD. In Hicksian terminology, BD is the "compensating surplus" for the price fall.

The above has described the situation of willingness to pay for reductions in probability of mortality. We now turn to the case of willingness to be compensated for increases in probability of mortality.

Increases in Mortality--

We again refer to Figure 10 to discuss the case where the price of safety rises. This is the situation which exists when an employer asks an employee to move to a riskier job. The individual would then have to pay more dollars to obtain the same level of safety.

The individual has an income of OM_2 and is initially faced with a price ratio of P_b. He is at equilibrium at point B and obtains OS_b of safety. When the price of safety rises to P_a the individual maximizes his utility where MRS = P_a (point A) and obtains OS_a of safety. In the process he has become worse off, moving to a lower indifference curve and lowering his consumption of safety. How much worse off he is can be investigated by constructing a line (M_1Z) with the slope of the old price ratio (P_b) and setting it equal to the MRS of the new indifference curve (U_1). This portrays the case where the individual's income is cut while at the same time holding the price constant. At the new point of intersection (C) he is as bad off as he is at A, yet the fall in utility from B to C is entirely the result of a fall in income. This amount, M_2M_1, is called the "equivalent variation" for the price rise, since it is the amount the individual's income would have to fall in order for his utility level to be equivalent to that obtained with a price rise alone.

The decline in consumer surplus due to the rise in price of safety can likewise be shown with the aid of Figure 10. The price rise results in a decline in the amount of safety purchased. The loss in net benefit to the individual is given by the vertical distance between indifference curves U_2 and U_1 at the old quantity. If BD is taken from the individual's income, he is returned to the same indifference curve that he obtains with the price rise. The amount represented by BD equals the amount which is lost from his income and which lowers him to the new indifference curve at the old quantity. This amount is called the "equivalent surplus" for the price rise.

It has been shown that the compensating variation for the price fall (the case of improved opportunities for safety) is equal to the equivalent variation for the price rise (the case of reduced opportunities for safety). The compensating variation for the price fall (M_2M_1) is the monetary measure of how much the individual must be compensated in order to entice him into the higher mortality rate situation embodied in the change of price from P_b to P_a.

For the purpose of the benefit-cost procedure, the appropriate measure of gain or loss is the consumer surplus involved in the price change. For a price fall, the compensating surplus (BD) is the gain in satisfaction that the individual derives due to the higher quantity of safety obtainable. For a price rise, the equivalent surplus (again BD) is the individual's loss in satisfaction due to the lower quantity of safety obtainable. As a practical matter, for small price changes, M_2M_1 is virtually identical to the consumer surplus measure needed in the benefit-cost analysis. This concept can be illustrated with the aid of Figure 10. As the price changes embodied in P_a and P_b become smaller (the slopes of the price lines anchored at income level M_2 approach each other), the difference (ED) between the appropriate variation (M_2M_1) and the appropriate surplus (BD) becomes smaller. As ED approaches zero, the surplus measurement (BD) approaches the variation measurement (M_2M_1).

How small a change in price is necessary to allow the assumption that M_2M_1 closely approximates the theoretically correct surplus measurement (BD)? This cannot be precisely defined since the two values are asymptotically approaching each other and will only equal each other when the slopes are the same (i.e., the price is unchanged). However, for purposes of this procedure, it is felt that for changes in the price of safety associated with changes in mortality rates below 1 in 10,000 (.0001 change in the probability of death), M_2M_1 approximates the consumer surplus involved (see Mann (31) for a discussion of individual's perception of risk).

Once it is established that the amount an individual is willing to pay for small changes in mortality rate is the correct measure of the change in consumer surplus, the question now arises as to how the monetary amount of a person's willingness to pay is found. Two approaches have theoretical validity: compensation for risk-taking and questionnaires of willingness to pay.

Compensation for Risk-Taking--

Compensation for risk-taking as a technique for estimating WtP_i has the advantage of being observable in the marketplace--specifically in the risky-job example cited earlier. If the individual agrees to undertake a hazardous occupation which increases his probability of death in exchange for a given sum of monetary compensation, that sum can be used as an estimate of his willingness to pay for safety. That incremental amount presumably reflects his indifference map in that it returns him to the higher utility curve he occupied before he took the riskier job. Thaler and Rosen (48) used data which measures the relative riskiness of jobs to estimate the trade-off between wages and risk. Thirty-seven broadly defined job classifications shown to be actuarially riskier than the average occupation were matched against a cross-sectional earnings survey. The results of their regression indicate that individuals in these risky occupations were willing to be compensated approximately $200 a year (in 1967 dollars) to incur an extra one in a thousand risk of death.

Thaler and Rosen point out that there are several reasons to believe that this estimate is conservative, especially when applying it to the general population. The occupations surveyed in their study were approximately five times riskier than the average U.S. occupation. People who take these jobs have different reservation prices for risk than the individual who picks a much less risky job. To use the derived estimate for the case of risk-averting individuals places a severely conservative bias on the aggregate willingness to pay. A further rationale for believing this estimate to be conservative is that the riskiness of these jobs can in some measure be affected by the individual employee. He thus is to some degree in control of his personal risk level within the context of the risk level of his occupation. If the individual is not in control of his personal risk exposure (for example, the person far removed from the pesticide application site who ingests pesticide residue in the food he eats), it is quite likely that he would need to be compensated to a greater extent to undertake the unwanted risk. This consideration is graphically illustrated by some people's willingness to accept a relatively high probability of death (.25 in a thousand) in an automobile accident while refusing to accept relatively low probabilities of death (on the order of 1 in a million) due to a reactor break-down at a nuclear power plant.

Questionnaires--

The other technique for assessing willingness to pay is the questionnaire method. It is widely recognized that this method has two basic problems. The first is the "free-rider" aspect, in that the individual cannot be excluded from the market. As a member of society he will derive benefits based on the willingness to pay responses of others if the project results in improved safety for all. The understatement of consumer surplus is likely to be large if it is generally believed that the project will be instituted regardless of the response to the questionnaire. This leads to the second problem, that of assuring unbiased answers. If the respondent feels he will actually be charged for the improved conditions according to his response to the questionnaire, self-interest provides a powerful incentive to understate his true willingness to pay. The necessity of couching the survey in terms with an immediate impact on the individual ("How much would *you* be willing to pay?") makes this the major conceptual problem of the survey technique. In spite of there difficulties, a methodology for determining willingness to pay for small changes in human mortality using the questionnaire technique has been developed by Mann, et.al. (31).

The results of the survey are somewhat disappointing since they diverge substantially from findings of previous studies. While there is no way of being sure of true risk preferences, the market derived estimate obtained by Thaler and Rosen (48) of \$200 for a 10^{-3} increase in risk is much lower than that found in the Mann survey. On the other hand, the survey indicated that individuals were able to answer the type of question posed, and that with an improved survey instrument, the willingness to pay questionnaire may be a workable concept.

Because of the limited population surveyed and the need to further develop the questionnaire, it is inappropriate to extrapolate results of the Mann survey to the general population. Until estimates are made on the willingness to pay for small changes in mortality by the general population, the results of the Thaler and Rosen occupational-risk study will be used in further calculations.

HUMAN HEALTH EFFECTS OF ALDRIN ON CORN: A CASE STUDY EXAMPLE

The insecticide aldrin is primarily used as a treatment to control soil insects which attack corn. Aldrin is rapidly metabolized into dieldrin which is itself a commercial pesticide product, although much less widely used. Since aldrin degrades so quickly into dieldrin, the remainder of this section will consider the acute toxicity effects of aldrin and the chronic effects of dieldrin on human health. The assessment follows the form described in Figure 7.

We begin the health effects section with the question of whether any health problems can be attributed to aldrin use. Certainly aldrin can be acutely toxic. High dosages of aldrin affect the central nervous system resulting in ataxia, tremors, respiratory failure and death. Ingestion of from 1 to 3 grams (g) of aldrin will cause severe poisoning symptoms (11). Consumption of such large dosages is restricted to occupationally exposed workers who come in contact with high concentrations of the active ingredients. These high concentrations, combined with aldrin's ability to be absorbed into the skin, give rise to the possibility of acute poisoning. However, instances of death due to aldrin poisoning in humans are quite rare and are the result of gross mishandling of the pesticide (23). The probability of death due to aldrin poisoning for a member of the non-occupationally exposed group is exceedingly small.

The attribute which makes aldrin an attractive agricultural pesticide--the ability to persist for a long period of time--is cause for concern when related to human health. In addition, it is lipophilic; that is dieldrin has a strong affinity for fat and is fat soluble. These two factors combine to raise the question of what effect, if any, does the storage of a long-lived metabolite of aldrin (dieldrin) in adipose tissue have on human health?

The answer to this question may be obtained either through epidemiological studies or through laboratory experimentation. The problems inherent in the epidemiological method have been detailed in the section headed "Chronic injury" (p. 54). Briefly, the lack of controls, the usually small sample size being examined, the relatively short time period over which a study exists, and the potentially irreversible damage to humans if the pesticide does result in a health effect, combine to make the epidemiological method a poor approach to use to discover if the chemical does cause health problems. It was on these grounds that an epidemiological investigation of aldrin by Jager (22) was found to be inadequate.

The laboratory approach has been criticized on the grounds that animal testing does not accurately reflect human susceptibility to chemicals. While this point is arguable, it is in fact true that, except for arsenic, all chemicals which have been shown to be oncogenic in humans have also demonstrated their oncogenicity in laboratory animals. On at least one occasion, laboratory results of oncogenicity presaged the equivalent finding in humans. The discovery that exposure to polyvinyl chloride gas caused a rare form of liver cancer in laboratory animals and man (6) lends credance to the use of laboratory testing for screening purposes.

In a laboratory experiment using CF-1 mice, Walker, Thorpe, and Stevenson (51) investigated the effect of dietarily ingested dieldrin. The experimental concentrations ranged from 0.1 part per million (ppm) to 20 ppm and the test periods lasted up to 132 weeks. A statistically significant increase in tumors was found at all concentration levels. The conclusion that may be drawn is dietary dieldrin does cause liver tumors in mice at the dose levels tested. The potential for health problems in humans exists and the analysis should proceed.

To ascertain the "tolerable" dose level for dietary dieldrin, a statistical analysis of the Walker, Thorpe, and Stevenson data is performed. The results of a pair of reports prepared by Mantel (34) (35) are given in Table 12. The reason for the choice of the probit model as well as the rationale for the selected levels of the variable parameters are given in the section headed "Mathematical Extrapolation." Briefly, it is felt that the choice of a slope greater than one and/or a confidence interval smaller than 99 percent results in an inadequately conservative estimate of the dose-risk relationship. (The dose-risk relationships for alternative parameters are also given in Table 12 for illustrative purposes.) The choice of "tolerable" risk has been set at 10^{-8} (1 in 100,000,000); this number being not too far from the intent of the "Delaney Clause."

Statistical extrapolation indicates that the dose associated with a 1/100,000,000 risk of contracting a liver tumor due to ingestion of dieldrin is 11 parts per trillion (ppt) per day in the diet. This "tolerable" dose level must be compared to the dose rate to which humans are exposed.

Because the laboratory experiment from which the extrapolation was drawn dealt in dietary exposure, it is correct to examine residues of dieldrin present in the human diet. The Total Diet Studies series conducted by the Food and Drug Administration is the best source of data on the average

TABLE 12. DOSE-RISK RELATIONSHIPS FOR DIETARY DIELDRIN USING THE MANTEL-BRYAN EXTRAPOLATION TECHNIQUE

	ASSOCIATED DOSE (ppt)			
RISK	SLOPE = 1.0 c.l. = 99%	SLOPE = 1.0 c.l. = 95%	SLOPE = 1.5 c.l. = 99%	SLOPE = 1.5 c.l. = 95%
10^{-8}	11	14	560	650
10^{-7}	26	36	1090	1220
10^{-6}	81	99	2160	2430
10^{-5}	250	300	4560	5140
10^{-4}	774			
10^{-3}	2430			

SOURCE: Mantal (35)

residues of dieldrin in food. The residue levels found in the survey are associated with the food consumption of a 15-20 year old male (see Table 13). This composition was chosen in order that residue estimates would be made on a high consumption diet, lowering the probability that the dietary exposure to pesticides would be underestimated. In the strict sense, the only individuals who have residue intakes as high as those found in the survey are those on similar high consumption diets. However, in the case of dieldrin, it is only necessary that a higher proportion of a smaller diet be made up of dairy products and meat in order to receive dosages of the same order of magnitude as found in the survey.

There is some evidence that, in the case of dieldrin, the diet survey gives a conservative estimate of total exposure, in spite of the attempt to make a liberal estimate of residue exposure by using high consumption diets. Robinson (44) investigated the relationship between dietary intake of dieldrin and the concentration of dieldrin in human adipose tissue and blood. The relationship between total intake and residues was found to be:

TABLE 13. RELATIVE COMPOSITION OF DIET BY FOOD CLASS

FOOD GROUP	AVERAGE GRAMS PER DAY	PERCENT OF TOTAL DIET
1. Dairy	769	22.6
2. Meats	273	8.0
3. Grain and Cereal	417	12.3
4. Potatoes	200	5.9
5. Leafy Vegetables	63	1.9
6. Legume Vegetables	72	2.1
7. Root Vegetables	34	1.0
8. Garden Fruits	89	2.6
9. Fruits	222	6.5
10. Oil and Fats	51	1.5
11. Sugars and Adjuncts	82	2.4
12. Beverages	1130	33.2
TOTAL	3402 g	100.0%

SOURCE: FDA Total Dietary Study (12).

Average total equivalent intake of
dieldrin in micrograms/day (ug/day) =

$$\frac{\text{mean (geometric) concentration of dieldrin in adipose tissue (ug/g)}}{.0185}$$

or

$$\frac{\text{mean (geometric) concentration of dieldrin in blood (ug/ml)}}{.00086}$$

The EPA's Human Monitoring Program is the source for data on U.S. human adipose tissue residues. In fiscal year 1970 the nationwide geometric mean was found to be 0.18 ug/g (25). Using Robinson's procedure, this translates into an average total equivalent intake of dieldrin of 9.73 ug/day. This compares with the 1970 average daily intake found in the Total Diet Study of 5 ug/day (9, p. 330). Since the diet is the dominant source of dieldrin exposure (when compared to respiration or absorption) for the general public, the diet study figure is probably low; how low cannot now be answered. To allow for this we assume that the figures for dieldrin found in the Total Diet Study are representative of the average dose per day in food for the entire population of the U.S.. This results in a conservative estimate of the dosage of dieldrin to which the general population is exposed.

In the fiscal year 1973 survey (15), typical market baskets of food were collected in 30 U.S. cities and examined for pesticide residues. The 30 composite average for dieldrin is given in Table 14. The major sources of dieldrin in the diet (on a ug/kg of body weight basis) are the dairy and meat groups. As an indication of the ubiquitous nature of the pesticide, twenty-six of the thirty dairy composites and twenty-nine of the thirty meat composites contained measurable residues of dieldrin. These two food groups accounted for 81 percent of the total average dieldrin residues found in the survey. The grain and cereal, potato, garden fruits and oil and fats groups each contributed minor amounts to the total.

As this is an examination of aldrin on corn, some assessment of the proportion of the residues which can be attributed to corn usage must be made. Aldrin on corn accounts for the vast majority of the pounds of active ingredient applied, and after subtracting the pounds committed to uses not associated with movement of residues (soil treatment for termites and treated hot caps, for example), the percentage devoted to the corn rises further. The assumption made in this analysis is that all dairy and meat residues are connected to the soil-insect treatments on corn. This assumption follows from the fact that most of the dairy and meat products in the U.S. either originate in geographic areas where aldrin is applied to corn (where the animals may be exposed to the pesticide directly through the air) or the animals are fed corn and fodder which has been treated with aldrin. The residues found in the other food groups would be difficult, though not impossible, to link to aldrin use on corn.

TABLE 14. RESIDUES OF DIELDRIN BY FOOD CLASS

FOOD CLASS	ppm	ug/kg/day
1. Dairy	.0016	.0180
2. Meats	.0037	.0147
3. Grain and Cereal	.0002	.0012
4. Potatoes	.0010	.0025
5. Leafy Vegetables	--	--
6. Legume Vegetables	--	--
7. Root Vegetables	--	--
8. Garden Vegetables	.0029	.0038
9. Fruits	--	--
10. Oil and Fats	.0001	.0001
11. Sugars and Adjuncts	--	--
12. Beverages	--	--
		.0403

SOURCE: FDA Total Dietary Study (16).

The result of this assumption is to set the average dietary intake of dieldrin residues due to corn usage at .0327 microgram/kilogram (ug/kg) of body weight/day. To convert this figure into parts per trillion of the diet, the following procedure is followed:

$$\text{Dose} = \frac{[\text{Residue in Food}]\ [\text{Mass of organism}]}{[\text{Quantity of Food}]}$$

$$\frac{[\frac{ug}{kg}]\ [kg]}{[g]} = \frac{ug}{g}$$

To express in parts per trillion: $\text{ppt} = 10^{-12}$, $\frac{ug}{g} = 10^{-6}$

$$\frac{\text{ppt}}{ug/g} = \frac{10^{-12}}{10^{-6}} = 10^{-6}$$

conversion factor becomes: $1 = 10^{6}\ \frac{\text{ppt}}{ug/g}$

$$\text{So, Dose} = \frac{[\text{Residue}]\ [\text{Mass}]}{[\text{Quantity of Food}]}\ [10^{6}]$$

Using data from Fiscal 1973 Total Diet Study,

Residue = .0327 ug/kg

Mass of organism = 69.1 kg (average mass of a teenaged boy)

Quantity of food = 2402 g/day*

$$\text{Dose (ppt)} = \frac{[.0327\ ug/kg]\ [69.1\ kg]}{[2402\ g/day]}\ [10^{6}] = 940$$

The average daily dietary consumption of dieldrin is 940 ppt. This is above the calculated "safe" dose and results in a finding that a potential human health problem does exist.

All previous discussion of residue levels was in terms of an "average" of the 30 composite market baskets examined. If the data is available, it is preferable to establish actual residue levels and the number of people associated with each level. As a practical matter, however, such a fine delineation between residue levels for the general population is not possible. The results of the Total Diet Studies show wide variations of residues in food groups obtained in the different sample cities. An attempt to fine tune the procedure by weighting the residue found times the population surrounding the sample city is an unwarranted use of the study. It is best to use the average figure, keeping in mind residue levels at any given location may be above or below the average. With this understanding, the estimation of the average risk is performed.

---

*The weight of the average diet (3402 grams/day) is reduced by the quantity of water included in the average diet in order to maintain comparability with the animal experiment from which the extrapolation is performed. This results in an average diet of 2402 grams/day excluding water.

The equation which may be used to convert dose into the upper limits of risk depends upon the variable parameters. With the slope equal to one and the confidence level equal to 99 percent, the equation (derived from data in (33) and (34) is closely approximated by:

$$\text{ppt} = (77 \times 10^{3})\, R^{1/2} + 3.3$$

where: R is the risk level, $R \neq 0$

ppt is the dose level

Under the probit extrapolation technique (with the variable parameters set as previously described) the 940 ppt dose translates into an upper limit to risk of 1.5×10^{-4} (1.5 in 10,000). An individual of the U.S. faces a risk as high as 1.5 in 10,000 of contracting tumors due to the ingestion of dieldrin residues resulting from the use of aldrin on corn.

The value attached to reducing this risk to the "tolerable" risk of 10^{-8} is estimated in the following manner.

Recognizing that an individual accepts higher risks up to the point where the compensation equals the expected value of the higher risk, Thaler and Rosen (47), as previously discussed found a willingness to be compensated for higher risk jobs of $200 per year per an increased risk of 10^{-3}. Assuming a linear extrapolation,

$$\begin{aligned}
\text{compensation} &= (\text{value of life}) \quad (\text{risk level}) \\
\$200 &= \quad V \qquad (10^{-3}) \\
\$2 \times 10^{5} &= \quad V
\end{aligned}$$

The evaluation under consideration in the dieldrin example is a movement from a higher risk level to a lower risk level--namely, 1.5×10^{-4} to 1×10^{-8}. Accordingly, the equation becomes:

$$\begin{aligned}
\text{compensation} &= (V) \ (\text{risk level \#1}) - (V) \ (\text{risk level \#2}) \\
&= (\$2 \times 10^{5}) \ (1.5 \times 10^{-4}) - (\$2 \times 10^{5}) \ (1 \times 10^{-8}) \\
&= \qquad \$30 \qquad - \ \$.002 \\
&= \$29.998
\end{aligned}$$

Thus, an individual with the given risk preference and facing the above change in probability will be willing to pay[a] $29,998 to attain the lower risk. Taking 214 million individuals to be the approximate U.S. population, the aggregate willingness to pay to avoid liver tumors connected with aldrin use on corn is $6,420,000,000. This estimate is in terms of 1967 dollars. A conversion to current dollars should be made. If the consumer price index is used as a measure of the change in the value of a dollar, the aggregate willingness to pay (in 1975 dollars) is $10,349,000,000.

The change in consumer surplus ($10,349,000,000), is the figure which enters the benefit-cost procedure from the human health section and is compared with the change in consumer surplus from the production section.

---

[a] The equivalence between willingness to be compensated and willingness to pay is shown in sections headed "Decreases in Mortality" (p. 66 and "Increases in Mortality" (p. 66).

BIBLIOGRAPHY FOR HUMAN HEALTH CHAPTER

1. Brown, J. D. Effect of a Health Hazard Scare on Consumer Demand. American Journal of Agricultural Economics, 51(3): 676-679, August 1969.

2. Cicchetti, Charles J. and A. Myrick Freeman III. Option Demand and Consumer Surplus: Further Comment. Quarterly Journal of Economics, 85(3), August 1971.

3. Clark, John Maurice. The Costs of World War to the American People. Yale University Press, New Haven, 1931.

4. Conley, Brian C. The Value of Human Life in the Demand for Safety. American Economic Review, 66(1): 45-55, March 1976.

5. Cranmer, Morris F. Reflections in Toxicology. Journal of the Washington Academy of Sciences, 64(2): 158-179, 1974.

6. Creech, J. L., and M. N. Johnson. Angiosarcoma of Liver in the Manufacture of Polyvinyl Chloride. Journal of Occupational Medicine, 16: 150-151, 1974.

7. Darby, William. The Philosophy of Acceptable Risk and the Practicality of Maximum Safety. In: Environment and Quality of Food, P. L. White and Dianne Robbins, eds. Futura Publishing Company, Mt. Kisco, N. Y., 1974.

8. Dublin, Louis I. and Alfred J. Lotka. The Money Value of a Man. Ronald Press, 1946.

9. Duggan, R. E. and P. E. Corneliussen. Dietary Intake of Pesticide Chemicals in the U.S. (III), June 1968-April 1970. Pesticides Monitoring Journal, 5(4), March 1972.

10. Edwards, W. F. Economic Externalities in the Agricultural Use of Pesticides and an Evaluation of Alternative Policies. Ph.D. Thesis, University of Florida, 1969.

11. Epstein, Samuel S. and Marvin S. Legator. Mutagenicity of Pesticides. The MIT Press, Cambridge, Mass., 1971.

12. Fein, Rashi. Economics of Mental Illness. Joint Commission on Mental Illness and Health Monograph Series No. 2. New York: Basic Books, Inc., 1958.

13. Finkelstein, Harold. Preliminary Air Pollution Survey of Pesticides, A Literature Review. Prepared for Public Health Service, U.S. Department of Health, Education and Welfare, October 1969.

14. F.D.A. Advisory Committee on Protocols for Safety Evaluation. Panel on Carcinogenisis Report on Cancer Testing in the Safety Evaluation of Food Additives and Pesticides. Toxicology and Applied Pharmacology, 2: 433, 1971.

15. F.D.A. Total Diet Studies: Fiscal Year 1973, Compliances Program Evaluation. Bureau of Foods, Unpublished Report, January 9, 1975.

16. Feldstein, Martin S. Distributional Preferences in Public Expenditure Analysis. In: Redistribution Through Public Choice, Hochman and Peterson, eds. Columbia University Press, 1974.

17. Fromm, Gary. Civil Aviation Expenditures. In: Measuring the Benefits of Government Investment, R. Dorfman, ed. Washington, D.C.: Brookings Institute, 1965.

18. Headley, J. C. and J. N. Lewis. The Pesticide Problem: An Economic Approach to Public Policy. Johns Hopkins Press, Baltimore, Md., 1967.

19. Hicks, J. R. A Revision of Demand Theory. Oxford University Press, 1956.

20. Hirshleifer, J., T. Bergstrom and E. Rappaport. Applying Cost-Benefit Concepts to Projects Which Alter Human Mortality. School of Engineering and Applied Science, U.C.L.A., Los Angeles, California, 1974.

21. Hoover, Robert and Joseph F. Fraumeni, Jr. Cancer Mortality in U.S. Counties with Chemical Industries. Environmental Research, 9: 196-207, 1975.

22. Jager, K. W. Aldrin, Dieldrin, Endrin and Telodrin: An Epidemiological and Toxicological Study of Long-Term Occupational Exposure. Elsevier Publishing Company, Amsterdam, 1970.

23. Kazantzis, G., A. I. G. McLaughlin, and P. F. Prior. Poisoning in Industrial Workers by the Insecticide Aldrin. British Journal of Industrial Medicine, 21(46), 1964.

24. Klarman, Herbert E. Syphilis Control Programs. In: Measuring the Benefits of Government Investments, Robert Dorfman, ed. Washington, D.C.: Brookings Institute, 1965.

25. Kutz, F. W. *et al.* Pesticide Residues in Adipose Tissue of the General Population of the United States, Fiscal Year 1970. Bulletin of the Society of Pharmacological and Environmental Pathologist, 2(3), March 1976.

26. Lave, L. B. and E. P. Seskin. Health and Air Pollution: The Effect of Occupation Mix. Swedish Journal of Economics, 73(1): 76-95, March 1971.

27. Lave, Lester and Eugene P. Seskin. Air Pollution and Human Health. Science, 169: 723-732, August 1970.

28. Lave, Lester and Warren Weber. A Benefit-Cost Analysis of Auto Safety Features. Applied Economics, 2: 265-275, 1970.

29. Lawless, E. W. *et al.* The Pollution Potential in Pesticide Manufacturing. Pesticide Study Series No. 5, Office of Water Programs, Environmental Protection Agency, June 1972.

30. Lederberg, J. 'Forward' to S. Epstein and M. Legator. The Mutagenicity of Pesticides. MIT Press, Cambridge, Mass., 1971.

31. Mann, Stuart H., Thomas J. Douglas, Jack L. Nasar and F. Roger Tellefsen. The Determination of the Economic Effects of Pesticide Use on Human Health: A Gaming Approach to Willingness-to-Pay. The Center for the Study of Environmental Policy Working Paper Series, No. 32, The Pennsylvania State University, University Park, Pa., 1976.

32. Mantel, Nathan and Marvin A. Schneiderman. Estimating 'Safe' Levels-- A Hazardous Undertaking. Cancer Research, 35(6): 1379-1386, 1975.

33. Mantel, Nathan, R. Bohidar Neeti, Charles C. Brown, Joseph L. Ciminera, and John W. Turkey. An Improved 'Mantel-Bryan' Procedure for 'Safety' Testing of Carcinogens. Cancer Research, 35(4): 865-872, 1975.

34. Mantel, Nathan. Estimating Limiting Risk Levels From Orally Ingested DDT and Dieldrin Using an Up-Dated Version of the Mantel-Bryan Procedure. Prepared for Environmental Protection Agency, April 1974.

35. Mantel, Nathan. Estimating Limiting Risk Levels From Orally Ingested DDT and Dieldrin Using an Up-Dated Version of the Mantel-Bryan Procedure. Supplementary Report, Prepared for Environmental Protection Agency, December 1974.

36. Mantel, Nathan and W. Ray Bryan. 'Safety' Testing of Carcinogenic Agents. Journal of the National Cancer Institute, 27(2): 455-470, 1961.

37. Mishan, E. J. Economics for Social Decisions. Praeger Publishers, Inc., 1973.

38. Mishan, E. J. Cost-Benefit Analysis. Unwin University Books, 1972.

39. Mishan, E. J. Pangloss on Pollution. Swedish Journal of Economics, 73(1): 113-120, March 1971.

40. Mishan, E. J. The Postwar Literature on Externalities: An Interpretative Essay. Journal of Economic Literature, March 1971.

41. Mishan, E. J. Evaluation of Life and Limb: A Theoretical Approach. Journal of Political Economy, 687-705, July-August 1971.

42. Mishan, E. J. Welfare Economics: An Assessment, North Holland, Amsterdam, 1969.

43. Mishan, E. J. Reflections on Recent Developments in the Concept of External Effects. Canadian Journal of Economics and Political Science, 31: 3-34, 1965.

44. Robinson, J. The Relationship Between the Dietary Intake and the Body Burden of HEOD (Dieldrin). European Journal of Toxicology, 2(4): 210-211, 1969.

45. Rice, Dorothy P. Estimating the Costs of Illness. American Journal of Public Health, pp. 426-44 , March 1967.

46. Schelling, Thomas. The Life You Save May Be Your Own. In: Problems in Public Expenditure Analysis, S. Chase ed. Washington, D. C.: Brookings Institute, 1968, pp. 127-176.

47. Smith, Wayne E. Factors Associated with Age-Specific Death Rates, California Counties, 1964. American Journal of Public Health, 58: 1937-1949, October 1968.

48. Thaler, Richard and Sherwin Rosen. The Value of Saving a Life: Evidence from the Labor Market. NBER Conference on Income and Wealth, 1973.

49. U.S. Department of Health, Education and Welfare. Report of the Secretary's Commission on Pesticides and their Relationship to the Environmental Health. Washington, D.C., December 1969.

50. Usher, Dan. An Imputation to the Measure of Economic Growth for Changes in Life Expectancy. In: The Measurement of Economic and Social Performance, M. Moss ed. Studies in Income and Wealth, No. 38, National Bureau of Economic Research, 1973.

51. Walker, A. I. T., E. Thorpe and D. E. Stevenson. The Toxicology of Dieldrin (HEOD). Food and Cosmetic Toxicology, 11: 415-432, 1972.

52. Warnick, S. L. and J. E. Carter. Some Findings in a Study of Workers Occupationally Exposed to Pesticides. Archives of Environmental Health, 25(4), 1972.

53. Weisbrod, Burton A. Economics of Public Health, Measuring the Economic Impact of Diseases. Philadelphia: University of Pennsylvania Press, 1961.

54. Winch, D. M. Analytical Welfare Economics. Penguin Books, Middlesex, England, 1971.

55. Zeckhauser, Richard. Procedures for Valuing Lives. Public Policy, 23(4), 1975.

CHAPTER 5

ENVIRONMENTAL IMPACTS*

INTRODUCTION

The use of pesticides has saved countless lives by controlling human disease vectors and by increasing the yields of agricultural endeavors. However, there is a growing consciousness that in addition to endangering human health the environment is becoming polluted by chemicals that can harm plants and animals as well.

This chapter is concerned with assessment of the effect of pesticides on plants and animals not raised for sale in the market place. Included under this heading are all manner of interrelated plant and animal associations and aesthetic considerations. It is in this area that a thorough investigation of the pesticide-caused effects will pay a dividend in the avoidance of "surprises" after initial registration or reregistration. By following chemicals, energy, etc. up the food chains, the basis for an impact identification and assessment scheme can be developed and, if desired, the impacts avoided. The inherent problems of pesticide-use are surveyed in this section and analytical tools are developed which will afford decision-makers a more realistic and comprehensive view of potential pesticide impacts on the environment.

First, the general effects of pesticide use will be delineated through discussion of the chemically induced alterations in the terrestrial sector (flora, soils and soil organisms, and higher trophic levels) and the aquatic sector (fresh water and esturine/marine flora and fauna).

Second, we will review some of the existing measurement techniques and illustrate why these schemes do not readily lend themselves for application to pesticide-use decisions.

This chapter relies primarily on descriptive material to examine pesticide impacts on the environment and aesthetic considerations. Quantification techniques commensurate to benefit-cost analysis, such as those set forth in the chapters on economic production and human health, do not exist at the present state-of-the-art for this chapter. Descriptive material is used, in lieu of superior approaches, to lay the basis for the development of a workable and conceptually correct evaluation technique for environmental and aesthetic impacts.

---

*Principal researcher in this section is Mr. Gary A. Shute, Center for the Study of Environmental Policy, The Pennsylvania State University. This section was prepared under the supervision of Dr. Donald J. Epp, Principal Investigator.

The development of an assessment technique consisting of adaptations of the Leopold matrix and Delphi method will then be presented. This section, through the use of food chain models and the evaluation matrix, will show the potential environmental impacts of pesticides based on the information presented earlier in the text.

Lastly, the evaluation procedure will be used with a specific example pesticide (aldrin-dieldrin) to further illustrate the envisioned use of the assessment scheme in actual pesticide-use decisions.

GENERAL REVIEW OF PESTICIDE MOVEMENT AND PROBLEMS

It is commonly acknowledged that the world cannot afford to abandon the use of pesticides. Crop production must be maintained and increased without significantly increasing the amount of agricultural land. Decreases in productivity due to pesticide restrictions cannot be fully offset by bringing marginal lands into production (11). A shift in demand towards high protein food as a country develops economically may bring about the discarding of former dietary habits; seriously impacting land use and pesticide-use decisions. Developing countries need cheap, effective pesticides to enable them to expand agricultural output (1) (47) (55) (61) (78) (81). Thus, pesticide residues will remain an acute environmental problem in the future (7) unless economically viable, low-residue pesticides or other pest control methods can be developed.

Even though pesticides have greatly benefited man, chemical and biological control agents employed to ameliorate factors that adversely affect man or his activities have their inherent problems. It is these imperfections that can cause deleterious effects in the biosphere.

Pesticides are formulated to kill or control a wide array of species considered harmful to man. The basic problem with these chemical formulations is that their effects are not limited to the intended or target organism. Through the use of pesticides, non-target organisms can be killed either directly or indirectly via the consumption of contaminated foodstuffs, plants, or animals. Pesticides can be transferred from their original application sites to other locations by erosion, drift, runoff and biological transfer.

The acute effects of pesticides, derived from a single exposure to the parent compound or its metabolite, are more readily observable than chronic effects. There are many natural factors affecting the uptake and accumulation of residues throughout the biosphere: choice, longevity, and formulation of the pesticide based on the pest to be controlled, and the target area's degree of isolation and climate. Formulation and application are greatly dependent on plants' abilities for translocation from the root or leaf (23) (88). In addition, particular plant species will differ in penetration, absorption and translocation between species and compounds. Soil

Insects

In reviewing the literature on the work that has been done on the routes of pesticide movement in insects, most experimentation has been confined to beetles which are predators or associated with decaying animal or vegetable matter. Three species of beetles (*Harpalus caliginosus*, *H. pennsylvanicus*, and *Poecilus chalcites*) were taken from a corn plot that had been treated with aldrin for fifteen years and analyzed for residues (62). The results of the analysis showed that *Harpalus* had 3.5 times the soil residue and *P. chalcites* had 31 times the soil residue and 9.6 times the residues found in *Harpalus*. The difference in residues was attributed to the fact that *P. chalcites* eat more animal matter that may already have residues whereas *Harpalus* eat mainly vegetation. This may be taken as an indication of biomagnification along the food chain.

Birds

The effects of pesticides on birds have been studied in great detail and include altered response to stress, altered liver function, altered vitamin content, abnormal testicular development, delayed ovulation, increased metabolism of steroids, failure to lay eggs, failure to deposit calcium in shells of eggs, decreased number of eggs, increased egg breaking and eating, inability to renest, decreased viability of the young, and increased tetratogenesis. More detailed information on pesticide residues and their effects on birds can be found in Edwards (26).

Several specific examples are worth noting. Ratcliffe (77) has reported widespread, rapid and synchronous declines in weight and thickness of shells of eggs layed by British peregrine falcons (*Falco peregrinus*) and sparrowhawks (*Accipiter nisus*). A causal chain between the phenomenon of frequent egg breakage, subsequent status of breeding populations and exposure to persistent organic pesticides has been made (77). Pesticides stimulated hepatic microsomal oxidation of drugs and microsomal hydroxylation of steroids (12). Bengalese finches (*Lonchura striata*) fed DDT had decreased fertility, hatachability and fledgling success (48). With pheasants (*Phasianus colchicus*) the feeding of near lethal doses of dieldrin had no apparent effect on that generation; however, the second generation of pheasants had significantly decreased hatchability and fertility, which seems to imply the possibility of genetic aberrations (3).

Laboratory experimentation is indispensable in studying the effects of pesticides on the environment, but it is of paramount importance that the laboratory findings be correlated with actual field observations. Most of the laboratory experimentation is concerned with the determination of LD_{50}'s and LC_{50}'s for various organisms in an attempt to indicate what the impact of a particular pesticide will be on the species in question.*

---

*LD_{50} is lethal dosage that results in 50 percent mortality of the test species; LC_{50} is lethal concentration in air, food or water resulting in 50 percent mortality.

characteristics and weather will markedly influence residue accumulation. The effects of residues and acute toxicities on plant species are also dependent upon plant metabolism.

Soil type also affects the availability of residues to both target and non-target organisms. If there is organic matter in the soil, there is less uptake of pesticides due to soil binding. For example, wheat, orchard grass, alfalfa, and corn had six times as much of a specific pesticide residue present when grown in sand than when grown in soil (97). Carrots grown in mineral soil with low applications of pesticides contained higher concentrations than carrots grown in organic soil with higher pesticide applications (41). Additionally, soils with clay colloids are highly significant sorbers of chemicals.

According to a study done by the Agricultural Research Service, USDA, under interagency agreement with the Office of Research and Development, EPA (93), even if there is heavy rainfall shortly after pesticide application the concentrations of the chemical in the runoff are very low--approximately five percent of the total application per year. However, this low amount of pesticide in the runoff is important since some chemicals are highly toxic to fish and other aquatic fauna and their persistence in the aquatic environment can be quite lengthy. In evaluating possible environmental impacts due to pesticide residue in runoff, toxicity and persistence must be considered together. When a pesticide residue is dissolved in water it can move from the application site and may not be tied into the sediment. Therefore, if runoff and erosion can be controlled, the probability of residue contamination in other areas will be lessened.

Several methods can minimize pesticide movement from agricultural application areas. These methods include:

- Adopt production techniques which do not use pesticides;
- The using of alternative pesticides that are not water soluble;
- Placement of pesticide with respect to movement;
- Optimizing pesticide formulation;
- Reducing excessive treatments;
- Optimizing the time of day for pesticide application;
- Optimizing the dates of pesticide application;
- Using lower pesticide application rates;
- Managing aerial application.

All of these measures are aimed at reducing the impacts of pesticides on the environment, but it should be noted that any one or combination of these control techniques may not be implemented due to physical or economic constraints.

In general, there are three major impact areas which encompass environmental effects: the terrestrial, the fresh water aquatic, and the esturine/marine aquatic environmental sectors. Our discussion will be focused on the first two of these sections since esturine/marine effects have not been as extensively documented. Environmentally there are basically three types of ecological consequences which have been identified (66). Pesticides can affect individual survival, reproductive success, and the genetic constitution of future generations.

EFFECTS OF PESTICIDES ON THE TERRESTRIAL ECOSYSTEM

Soil Invertebrates

Soil invertebrates can be readily affected by pesticides due to their proximity and contact with application sites (e.g., direct application to soil or drift and fallout from application to plants). There has been comparatively little work done in this area, but there are a few examples of the possible impacts pesticides could have on non-target soil invertebrates. Earthworms help to establish and maintain the structure and fertility of the soil, an activity with obvious agronomic importance (84). Therefore, increased earthworm mortality may affect soil fertility. In areas where DDT was used to control Dutch elm disease, residues in earthworms were eight times greater than in worms from nearby but untreated soil (45). Residues in earthworms are a function of earthworm species and habitat and residue concentration and stability. While there are numerous examples of the uptake and storage of pesticide residues by earthworms in forests, orchards, and arable land, residue concentration tends to be greater from forests and orchards than from arable lands. Other soil invertebrates that have been studied to date include molluscs (snails and slugs). Reports indicate that residue retention in slugs tends to be greater than in earthworms taken from the same site (91). A comparatively small amount of work has been done, however, and what has been done seems to suggest that the uptake mechanism by snails and slugs is similar to that of earthworms.

The more persistent pesticides may cause more sub-lethal effects (66). The extent of these sub-lethal effects is quite important. Earthworms, for example, become easier prey for birds which tend to accumulate residues themselves thus illustrating food chain transfer of pesticide residues and a possibility for biomagnification (18) (50). Sub-lethal doses of pesticides can also affect the predator-prey relationship in that the predator may now become prey due to altered behavior patterns (52).

However, it is extremely difficult for these *in vivo* studies to indicate food chain reactions successfully in the environment due to the absence of laboratory controls in the field. That is, single route exposures in the laboratory may not correctly represent polyvalent field exposure through dermal uptake, inhalation, drinking, and ingestion. Some field studies indicate that the breeding success of the woodcock *(Philohela minor)* in New Brunswick varied inversely with the amount of DDT used and with the total area sprayed (99). In Wisconsin, when elm trees were sprayed with greater than ten pounds per acre of DDT, the wild robin *(Turdus migratorius)* population had a greater than 85 percent spring mortality and, due to the loss of adults, virtually no reproduction (46). In addition, there was reported an increase in reproductive failure of Western grebes *(Aechmophorus occidentalis)* due to DDT at Cedar Lake, California (42). Poor reproduction in the golden eagle *(Aquila chrysaetos)* in Scotland was correlated to the amount of organochlorines ingested (59). The list of comparable examples is quite extensive and has been condensed to the above mentioned examples to afford some understanding of the possible consequences of pesticide use or misuse and the long term effects of pesticidal residues in the biosphere.

Mammals

One of the major sectors of the environment which can be affected by pesticidal agents is mammals. The impacts upon mammals will be considered in detail in the section on Aidrin-Dieldrin, and only two specific effects will be noted here. Frist, shrews *(Microsorex hoyi)* have been shown to have a particular propensity for residue accumulations (22). Second, mink *(Mustela vison)* accumulated greater DDT residues than the hares *(Lepus americanus)* on which the mink fed, once again indicating possible biomagnification. Examples of wild mammal mortality due to aldrin, dieldrin, and endrin can be found in Dustman and Stickel (24).

FRESH WATER AQUATIC EFFECTS

When exposed to pesticides, fresh water flora and fauna exhibit many of the same reactions as those discussed in the terrestrial section. In the case of fresh water flora, there was shown to be a decrease of 40-90 percent in phytoplankton *(Scenedesmus quadricauda)* production due to organochlorine exposure (76). Fish represunt the sector of the fresh water environment that has the greatest amount of research effort. Certain species of fish have demonstrated the ability to develop strains resistant to chlorinated hydrocarbons. These resistant nontarget organisms include some aquatic invertebrates and six species of fish and amphibians (29) (30). Resistant mosquito fish *(Gambusia affinis)*, when bred in a non-contaminated system, retained their resistance to the pesticide.

Resistance to pesticides can be attributed to a number of different factors, alteration of permeability at sites of action, changes in respiratory surfaces, increases in fat content, changes in the excretory system, detoxifying enzyme systems, novel metabolic pathways and rates, and changes in antibody reactions (25). The major problem afforded by the development of resistant strains of fish is that the resistant strain will carry greater body burdens to higher trophic levels. Resistance enables the adapted species to survive the environmental stress (e.g., pesticides) so that the survivors are physiologically prepared to cope with a particular stress.

Alteration of normal activity patterns has been noted in the Atlantic salmon *(Salmo salar)* which underwent changes in its temperature selection gradient (69). It has also been noted that pesticide exposure can alter the salinity preference of fish. One must consider what effect the alterations in activity patterns will have on the distribution and survival of the organisms.

ENVIRONMENTAL IMPACT ASSESSMENT

Where pesticides are used to a great extent there exist possibilities for synergistic or antagonistic interactions between different pesticides. Studies have shown that there was an interaction between dieldrin and DDT which affected the accumulation in pyloric caccae of rainbow trout *(Salmo gairdneri)*, so that DDT residues increased and dieldrin residues decreased over the amounts taken up from the amount of single insecticides (60).

There is a dual threat of pesticidal accumulation in fish. The first threat is that the residues in fish may exceed man's toxicity level (tolerance) but this has been judged unlikely. Secondly, pesticides could bring about the elimination of certain species that are an important source of food and recreation by exceeding threshold toxicities (25). One must be cognizant of the distinction between the elimination of a resource because of its physical decline (toxicologic impact) and the elimination of resource availability due to regulation (exceeding tolerance levels). In addition, secondary losses through increased mortality via reproductive failures and increased predation resulting from alterations of mobility and activity patterns is of the utmost importance.

Our understanding of nature's interrelationships is extremely limited. The tentative nature of human health effects noted in the previous section shows that man is only beginning to understand the process by which chemicals affect our own species. By comparison, our level of understanding of the effects of chemicals on the myriad of lifeforms in nature is in its infancy. Even within a related family of animals (rodents, for example), the effect of chemicals is likely to differ among the individual species.

Importance of Species Survival

If a species can be identified as threatened by the application of a pesticide, the assessment of the effect of its demise can begin. It is at this point that a thorough understanding of its place in the ecosystem is important, since its extermination will ripple through the environment as illustrated in the previous section. In general, there will be an impact on the organisms upon which it feeds--without the downward pressure of predation, the food organism may multiply greatly, causing its own impacts to change. (We see a small scale example of this effect when, in trying to limit the growth of a harmful insect, a pesticide inadvertently kills the target insect's natural enemies. Without the depressive effect of the predator, the target insect rapidly reestablishes itself, often at levels higher than before the spraying.) There will also be an impact upon the organisms which feed on the threatened species--removal of a food supply could lead to extinction of a higher predator, if it is a specialized feeder, or merely a temporary dislocation until other food sources can be utilized by a general feeder.

The concept of option demand, discussed in Chapter 4, is useful in measuring the importance of species survival. Once the species is gone the possibility of ever observing it has passed. Certainly the attributes which describe option demand existence are present; there is uncertainty about the future demand for the commodity, there is an element of irreversibility in the decision to supply a commodity in the future, and there is no way for the resource owner to exclude those who do not pay for the benefit (9). Thus if consumer surplus can ever be measured for environmental preservation, an additional amount will have to be added for option demand maintenance.

Measurement of Environmental Effects

This section examines the question of whether environmental effects of pesticides can be measured. The past decade has seen rapid growth of concern about the conditions of the natural environment. Just from looking at the complexity of the problem, one could say measurements would be quite difficult. Evaluation of man's impacts on the environment is still in its infancy. Unfortunately, present econometric techniques do not allow us to make the type of measurements we would like. We are dealing with a public good--the environment--all of which is not impacted in each pesticide-use decision.

Incorporation of environmental impacts into a benefit-cost or other framework is imperative. According to Howe the problems inherent in environmental measurement would be minimized "if there were only two ingredients to be considered, (a) infallible indices of environmental conditions and, (b) value weights that, when multiplied by the indices, would permit environmental values to be added in with the normal benefits and costs of the action under consideration." (44, p.26) Such a procedure is clearly not available at present, but this is not to say that environmental impacts should not be considered at all when attempting to evaluate the desirability of a particular action.

Much of the work to date has been confined to the area of water resource planning and evaluating the impacts of aesthetic considerations. Thus far, studies which have attempted to measure environmental effects quantitatively have usually used the expenditures approach. In this method, the expenditures of those who partake in the use of environmental resources are totaled and the sum is used as a proxy for the value of the wildlife involved. On a nationally aggregated basis, this technique may measure the value of goods and services consumed in the enjoyment of the wild. However, as a measure of the value of the wildlife, it has serious theoretical problems. Furthermore, the environmental impact of a pesticide is likely to impact on only a small number of animals or plants. Imputing a value from such a national survey would be impossible. Use of this technique on a local level would still only give a theoretically defective measure of aggregated wildlife value with no capability for species specificity.

There are numerous studies which attempt to measure the value of a specific outdoor activity--usually hunting or fishing. There is some question of what should be used as the value variable--the license fee paid to the state, the license fee plus travel costs of reaching the site, or the net of all costs over that which would have been spent if the activity was not engaged in. These studies do, however, answer questions of concern to hunting and fishing agencies.

The problem with relying on this method of valuation in pesticide matters is that often the threatened species is not hunted or fished. The implication of the expenditures method is that these species are not valued. If a species was once hunted but now has been placed on the protected list, is its value now zero? If it is protected, the species has value to someone, so clearly the answer to this question is no.

The previously described methodologies do not provide a complete assessment of environmental impacts in a manner which can be successfully integrated into a benefit-cost framework providing a comprehensive view of environmental consequences. In addition to these methods, there are numerous other schemes which attempt to incorporate environmental values into the decision-making process, but are mainly limited once again to water resource planning, aesthetic consideration, and highway construction planning. An evaluation of seventeen such measurement techniques can be found in "A Review of Environmental Impact Methodologies" by Warner and Preston (95). Most of the models are designed to be site-specific, such as the Dee (19) (20) techniques for water resource projects and the Little (57), McHarg (62) and Smith (87) techniques which are primarily concerned with the construction of transportation facilities.

It is unlikely that the application of pesticides will be confined to a relatively small area and thus the use of site-specific models does not possess the ability to account for all the possible environmental consequences that are attached to pesticide introduction into the biosphere. In fact, no such technique exists today, but it is quite clear that the environmental consequences of pesticide use cannot be ignored.

In reviewing the techniques available to date, it appears that a technique with less site-specificity and general overall flexibility would be the most appropriate. In an attempt to incorporate pesticidal impacts into a benefit-cost analysis, we have developed a methodology that is a meld of the Leopold matrix (56) and the Delphi technique (16). The Leopold matrix is a grid-structure interaction device in which pesticide effects and impacted organisms can be identified and then assessed on a relative qualitative and quantitative basis; with the perspective of the evaluator critical to assessment determination. The information obtained from the matrix analysis is then further scrutinized via the Delphi technique which consists of a multiround iterative examination of the problem culminating in a concensus, but at the same time, permitting individual differences of opinion to remain at the final decision point.

The Food Chain Model

We envision that the evaluator will begin the procedure by first examining the target organism and its place in the ecosystem. The first step of the evaluation procedure is a formidable task in itself. General food chain relationships are available in almost any biology textbook, but this particular evaluation scheme dictates that the interrelationships among organisms within the specific impact or pesticide-use area be examined quite closely. This is not to say that the environment as a whole should not be examined, but it is of the utmost importance that the evaluation of the pesticide-use area be as complete as possible from the onset so as not to omit other possible environmental ramifications.

Chemicals can be introduced into the environment during manufacturing, distribution (transport to application site, storage, etc.), application and disposal of unused portions of the formulation and/or the containers. The introduction mode will affect the impact on the environment and the point at which the analysis will begin. Regardless of the point of introduction, however, the thorough consideration of the initial effects of the pesticide on target and non-target organisms and the food chain relationships of these organisms is imperative.

The Model

Gillett, *et al.* (35) have collected a series of conceptual models illustrating the possible movement of pesticides through the environment. Figure 11 is a representation of the movement possibilities of a pesticide in the faunal sector of the environment. Upon entry into the system, a pesticide's possible movement can be traced through the model, thus affording information on organisms that may be affected. The impacted segments of the environment will then be examined in the evaluation matrix. Note that this faunal model is only a sub-section of the environment and the evaluator will be charged with the responsibility of examining other facets of the biosphere, such as the floral, atmospheric and fresh water and esturine/marine aquatic systems, in the same fashion. Each sub-section of the environment can be integrated with the other to provide a total summary such as illustrated in Figure 12.

The conceptual models can be modified to be more explicit for a particular chemical agent in a specific environment. This systematic scrutiny of the food chain interrelationships and pesticide movement through the environment will help to minimize the "surprises" after initial registration or re-registration of a pesticide. Through the use of these models, the evaluator will possess a more congruent picture of the physical, biological and chemical interrelationships between pesticides and the environment.

Closer examination of Figure 12 will afford better comprehension of the envisioned operation of the model which is divided into three major sections: atmospheric, terrestrial and aquatic. Each of these categories is further broken down into its integral parts such as, 1) Atmospheric--mesosphere, stratosphere and troposphere; 2) Terrestrial--soil divisions, fauna and flora and 3) Aquatic--marine, fresh water and esturine. More specifically

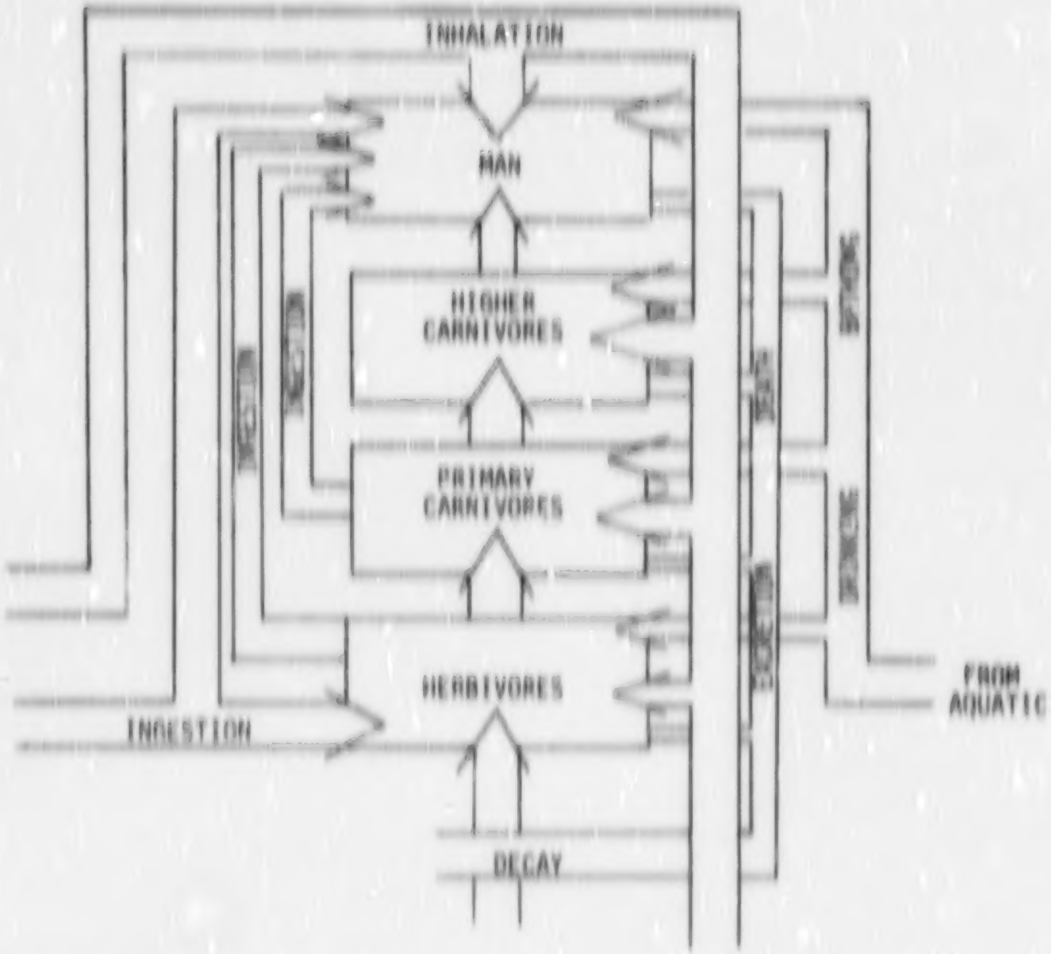

INHALATION
MAN
HIGHER CARNIVORES
PRIMARY CARNIVORES
HERBIVORES
INGESTION
DECAY
FROM AQUATIC

Figure 11. General pesticide movement model in faunal sub-system

[Adapted from Gillett, (35, p. 34)]

DEFINITION OF TERMS FOR FIGURE 11

Herbivore = A plant-eating animal, also considered to be primary consumer.

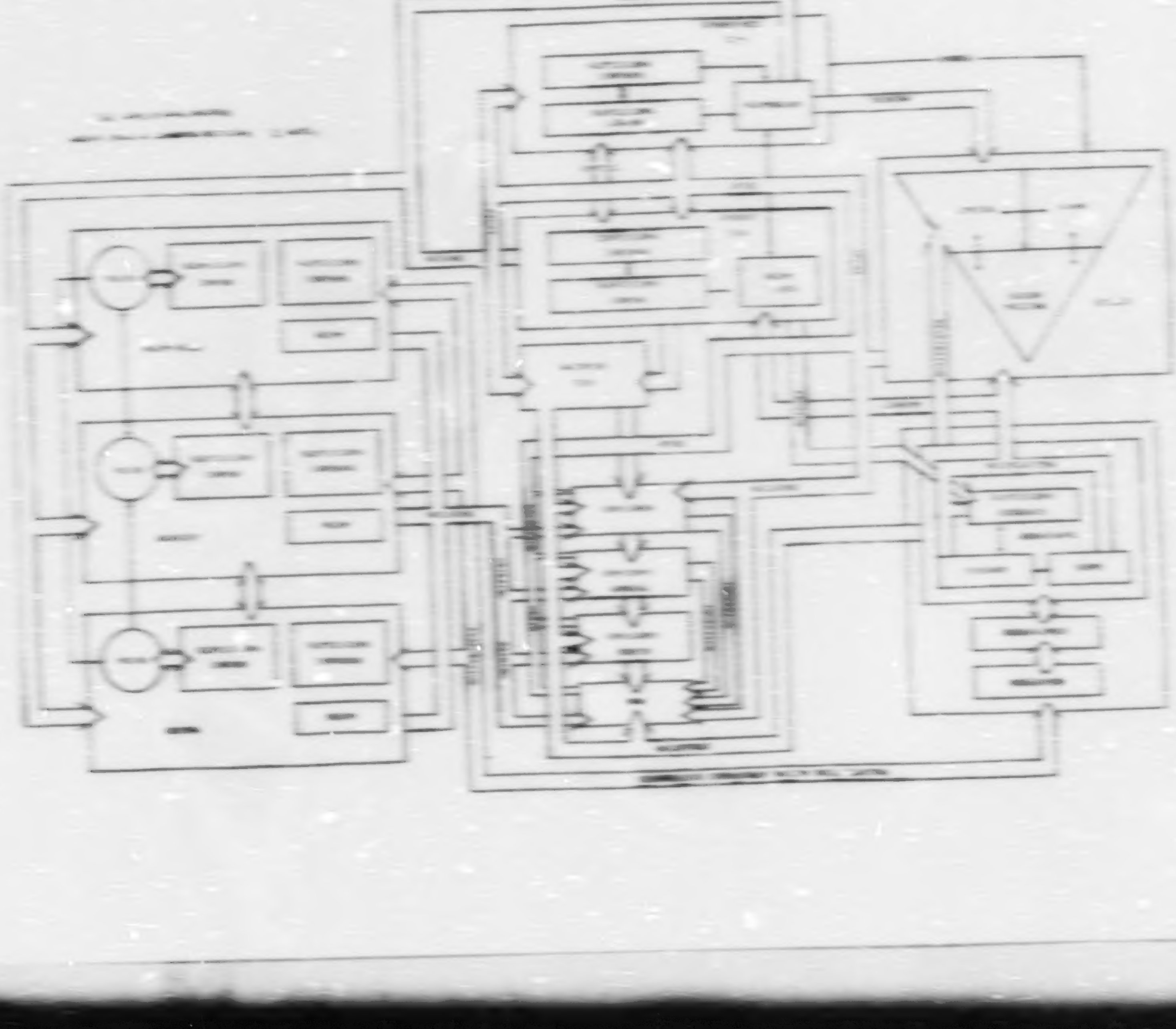
Carnivore = Any order *(Carnivora)* of flesh-eating animals. Animals preying upon primary consumers of primary production, e.g., predaceous mites feeding on phytophagous mites.

Decay = Decomposition of proteins chiefly by bacteria. Conversion of particulate organic carbon (POC) to dissolved organic carbon (DOC).

Ingestion = To take in for digestion by mouth.

Inhalation = Process by which animals and plant cells utilize oxygen, produce carbon dioxide and conserve the energy of food stuff molecules in biologically useful terms, such as ATP.

Excretion = Removal of metabolic wastes by an organism.



DEFINITION OF TERMS FOR FIGURE 12

Biota = The fauna and flora of an area, first order producers and consumers, second order consumers, higher consumers and omnivores.

Primary Producers = Green plants that manufacture organic compounds from simple inorganic substances.

Primary Consumers = Organisms that prey on the primary producers.

Decomposer Organisms = Bacteria and fungi which break down the organic compounds of dead cells from producer and consumer organisms into inorganic substances that can be used as raw materials by green plants.

Cuticle = A waxy layer that appears on the exposed surface of the cell.

Root = Sub-surface portion of a plant which is structurally adapted for the functions of absorption, anchorage or storage.

Fruiting Bodies = Structure which produces fruits which are seed-bearing structures derived from flowers.

Foliage = A cluster of leaves, flowers and branches.

Flora = The plant life characteristic of a region, period of special environment.

Fauna = The animals or animal life of a region, period or geological stratum adapted for life in a specified environment.

Mesosphere = A layer of the atmosphere extending from the top of the stratosphere to an altitude of approximately 50 miles.

Stratosphere = An upper portion of the atmosphere which is above approximately 7 miles, depending on latitude, season and weather and in which temperature changes little with changing altitude and clouds are rare.

Troposphere - The portion of the atmosphere which is below the stratosphere which extends outward about 7-10 miles above the earth's crust, and in which generally temperature decreases rapidly with altitude, clouds form and convection is active.

Particulates - Minute separate particles.

Omnivores - Organisms feeding on both plant and animal substances.

each of these sub-sections can be expanded to be even more explicit depending upon the needs of the particular evaluation. The atmospheric section of the environment, which has already been divided into the mesospheric, stratospheric and tropospheric zones, can be expanded to illustrate the interaction of gases, aerosols and suspended particulates in the troposphere which also is affected by fallout, washout, co-distillation and erosion.

The faunal sub-section of the terrestrial sector illustrates the interaction of herbivores, primary carnivores, higher carnivores and man. The floral sub-sector shows the interrelationships between plant cuticles, foliage, roots and fruiting bodies. Notice also the relationships of the soil divisions with one another, i.e., soil surface, soil sub-surface and soil organisms. Each of these divisions contain important interactions which can affect the fate of pesticidal movement and have deleterious effects in the environment. For example, the soil surface compartment contains components such as surface water, inorganic particulates, organic particulates, etc. which may affect the retention, mobility, degradation, and toxicity of a chemical agent. The same components are identifiable within the sub-surface soil. The entire soil complex is integrated into the environmental model through erosion (wind, mechanical, abrasion), decay of matter, imbibing or leakage of water and inhalation just to illustrate a few interaction mechanisms.

Examination of the aquatic sector once again shows a wide array of interdependencies that will affect the fate of chemicals in the environment. The marine, fresh water and esturine spheres all have common basic components that are crucial in tracing pesticide movements and effects, i.e., water, organic and inorganic particulates and the aquatic biota itself consisting of primary producers and consumers, secondary consumers, higher consumers and omnivores.

All of the environmental sectors interface in some manner, showing the essence of the pesticide problem. That is, the chemical formulation affects not only the target organism in the target area, but through these interfaces and food chain relationships, can be transported to other segments of the biosphere posing serious environmental ramifications. Once a pesticide's possible movements and effects are established through these types of food chain movement models, the impacts can then be transferred to the evaluation matrix for further examination.

DDT Example

It may prove advantageous at this time to trace the movement of the pesticide through the models to illustrate further the envisioned operation. Assume that DDT is sprayed to control Dutch elm disease. Barring any accidental introduction into the environment and not considering disposal of containers and unused formulation, the point of entry will mainly be in the floral sector of the environment with some introduction into the atmosphere. The DDT entering the atmosphere (mainly the troposphere) will be in the suspended particulate form and can be transported via wind movement to other locations. The atmosphere is a transport medium and the pesticide (DDT) will not do much until taken up by an organism even though it remains biologically active as a vapor or particulate and can react chemically or photochemically. Eventually, the DDT will be deposited on the aquatic and/or terrestrial segments of the environment with the concentration being diluted by dispersion. Deposition is usually via dry deposition, sedimentation and washout by rain. This serves to illustrate the possible movements of the DDT application from the site to other sectors of the environment in accordance with Figure 12. Note: However, this is not a statement as to whether the redeposited DDT is toxic, either acutely or chronically. There are many factors within the atmospheric sector that will affect the pesticide's movement and concentration which the evaluator must consider (96). This procedure traces only one path of a pesticide's possible movement. Now the evaluator must return to the point of entry (floral sector) and examine alternative routes of displacement.

Recalling that DDT was sprayed on the floral sector to control Dutch elm disease, the pesticide is now in direct contact with the foliage and cuticle. The DDT that does not adhere the foliage is deposited on the soil surface in which the pesticide can be introduced into the surface water and the inorganic and organic particulate complex. The compound can remain bound in this sector or can be transported to the soil sub-surface through percolation, to the aquatic sector through runoff, erosion and sedimentation, to soil organisms through ingestion and physical contact (absorption) and to the atmosphere via abrasion or volatilization. Soil organisms (e.g., earthworms) are ingested by primary carnivores (e.g., robins) in the faunal sector (63) (5). The primary carnivores are the prey of the higher carnivores leading ultimately to man. Therein exists the possibility of residue accumulation, biotransfer and biomagnification.* Dead organisms or excreta from the faunal sector (through residues) may threaten the aquatic, soil, atmosphere and floral sectors of the environment. Leaf drop from the floral sector can also add to the soil residue reservoir.

Examining the aquatic sector more specifically, surface runoff and aerial transport are the major vehicles for translocating pesticides into the aquatic ecosystem (70). The contaminant is then made available to the aquatic organisms in the dissolved or suspended particulate form. Primary producers (phytoplankton) have shown a 40-90 percent decrease in production due to DDT exposure (76). This will affect the primary consumers by decreasing the food supply and/or through the ingestion of residues. The aquatic food chain can be followed through to ascertain what other organisms may be affected. Contaminants from the aquatic sector can reenter the other sectors of the environment via evaporation and precipitation, ingestion of aquatic organisms by terrestrial organisms (possibly by man directly or indirectly) and by plants and animals imbibing contaminated water.

---

*Residue accumulation is a function of (a) input rate and (b) output rate (excretion and metabolism). When a > b, residues will accumulate. The critical point is the metabolism/excretion factor. Therefore, bioaccumulation is a consequence of biotransfer when a exceeds b with biomagnification occuring when the accumulation in the organism is significantly higher than concentrations in the air, food, or water (34).

We have shown that the tracing of a pesticide's movement through the environment is a formidable task, but it is also both possible and rewarding. Using the characteristics inherent to a particular pesticide, application information (i.e., application rate, frequency area, etc.), knowledge of existing or probable effects and food chain models such as the ones discussed, the deleterious impacts of pesticides on the environment can be determined. It has been illustrated that soil invertebrates can accumulate large residues of organochlorine insecticides. The same is true for beetles, molluscs and mites. Birds accumulate these residues when they ingest contaminated food stuffs and in turn are eaten by vertebrates on a higher tropic level. It is important to indicate however, that ecologically significant flows of carbon and energy will follow these same pathways whereas pesticide effects will be seen in the models where they tend to control (stimulate or inhibit) the flux along the pathways. Furthermore, each effect must be measured and quantified in dose-response terms in order to facilitate mathematical explicitness in modeling.

The Impact Matrix

Using the proposed type of model, the movement of a particular pesticide through the environment can be suggested and the organisms it may affect possibly detected. Once this procedure is complete an evaluation of the impacts of the pesticide on the environment can begin using the impact matrix. It is recognized that the model system is not absolutely all inclusive; however, a systematic approach such as this will help to insure that fewer environmental "surprises" will result.

It is possible that such a rigid procedural matrix may not be the most effective method of achieving the desired level of completeness, and therefore each of the categories of effects and impacted organisms will be left general subject to specific expension needs. When the expansion of these categories is complete, the matrix will have a detailed breakdown of effects and organisms affected with slash marks where the two intersect.

Figure 13 is an example matrix of the general impacts on the environment sector based on the information set forth in the section discussing the general impacts of pesticides on the environment. To briefly review the matrix as it stands before the assessment begins, each effect which has been found (listed vertically) has a potential impact on the environment (listed top horizontally). A slash will be placed at the point of intersection to mark this impact. Within each box a number in the upper left hand corner (from 1-10) which indicates the relative magnitude of the impact will be inserted by the evaluator. After this, the evaluator will place a number (1-10) in the lower right hand corner which indicates the relative importance of the impact (one represents a low importance or magnitude while ten represents high magnitude or importance). The term "magnitude" is used in the sense of degree, extensiveness or scale. The term "importance" is a relative measure of the significance of the impact and is based on the value judgments of the evaluator. (56)

To illustrate the procedure, suppose that the pesticide under consideration is linked to reproductive problems in California condors. If the pesticide is to be used in areas where condors rarely venture, it might be

possible to assess the magnitude of the impact as fairly low (say 1 or 2). However, if the importance of the condors' survival is deemed high enough, the importance might be assessed at a high level (say 9 or 10).

Conversely, if a pesticide is applied to a particular area and a substantial part of the soil invertebrate population (slugs and snails) is killed, the magnitude of the impact is high, but the relative importance of the impact may be deemed low or high, depending on point of view.

The impacts illustrated on the general matrix refer to no pesticide in particular and the evaluation numbers have been omitted at this time since no specific chemical agent or dosage is indicated. However, it is assumed that the inferred application rate is that which would be associated with the pesticide's particular use. Many of the impacts indicated in Figure 13 cannot be foreseen without the use of the food chain models developed previously. Only after an impact has occurred will it be entered into the matrix, or it may even be omitted if the particular event is not readily attributed to a particular pesticide's use.

An example may assist in clarifying this line of thought. Returning to the use of DDT to control Dutch elm disease, it was initially speculated that spraying operations might have some deleterious effects on birds in the spray area. Therefore, tests were carried out on animals in the proposed spray area in an attempt to ascertain the possible consequences. In doing this robins were sprayed with DDT and the effects were thought to be minimal. But, when the Dutch elms were sprayed, there was a dramatic decrease in the wild robin population. It was finally deciphered that the earthworms within the sprayed area, upon which the robins fed, contained DDT residues that were eight times greater than those found in earthworms from untreated soil. This experience serves to illustrate that examination of food chain relationships, such as those discussed in the previous material, is imperative *prior* to a pesticide-use decision. Through the use of such food chain models and environmental check lists, the number of environmental hazards posed by a pesticide-use decision will be kept to a minimum. It should be noted that the general pesticide impact matrix presented here is based only on those illustrations of pesticide effects presented in the text and therefore is by no means comprehensive. The abscissa and ordinate on the matrix are assumed to be "open-ended" in the sense that the evaluator is encouraged to enter additional effects or impacted organisms into the basic matrix as appropriate.

The Delphi Technique

Dalkey (16) added iteration and controlled feedback to a system designed for decision-making using group prediction. In general, this "Delphi" procedure has three main attributes: a) anonymity, b) controlled feedback, and c) statistical group response (15). Anonymity serves to suppress the influence of a dominant member of the response group. This is usually accomplished through the use of a questionnaire, thus eliminating face-to-face contact. This feature of Delphi lends itself well to the type of analysis we have envisioned in that where there is a lack of data, experts from various geographic areas can be called upon to provide input (even if subjective) for a pesticide-use decision. This is a highly desirable feature since each pesticide has inherent to it certain characteristics that will dictate its environmental and aesthetic impacts. Since it is unlikely that experts will be assembled at the decision-making unit for each pesticide-use inquiry, the Delphi technique is useful in that information input can be collected from a large geographic area by mail or telephone.

Controlled feedback is a device by which the results of previous assessment rounds are summarized and then communicated to the participants for further comment. This process will also tend to limit the amount of speculative or uninformed opinions entering the analysis. The evaluators could be given a basic (such as that in Figure 13) matrix at the onset and be asked to identify any effect and organism affected that they can and rate the impact for the particular pesticide under consideration. These results will be compiled and summarized and then redistributed to the evaluators for iteration. This process will continue until the chief investigators deem termination to be appropriate.

Finally, statistical group response is a way of reducing pressure for group conformity. This results in the possibility of widespread opinion at the end of the controlled feedback stage, permitting the representation of individual opinion in the final response.

In evaluating environmental and aesthetic impacts, the evaluators will be basing their numerical ratings primarily on value judgments, tending to obscure the validity of the procedure. However, if the respondents will accept statistical aggregation of weights supplied by the group, the procedure is still feasible (15, p. 73).

The Delphi technique is a rapid and efficient means to aggregate the views of knowledgeable personnel. When Delphi is integrated with the food chain models and environmental matrices, it will provide a relatively reliable representation of pesticide-use decision impacts on the environment.

ENVIRONMENTAL IMPACTS OF ALDRIN APPLIED TO CORN: A CASE STUDY EXAMPLE

The purpose of this section is to provide an example for use of the model presented in Figure 12 by tracing the movement of aldrin-dieldrin through the environment. Aldrin (1,2,3,4,10,10-hexachloro-1,4,4a,5,8,8a-hexayhdro-*exo*-1,4-*endo*-5,8-dimethanonapthalene) is readily broken down into dieldrin, at least 85 percent active ingredient HEOD. (1,2,3,4,10,10-hexachloro-6,7-expoxyl-1,4,4a,5,6,7,8,8a-octa-hydro-*exo*-1,4-*endo*-5,8-dimethanonapthalene). It is the latter compound that will be examined more closely so as to determine its movement through the environment and its possible affects.

The majority of aldrin used is in the agricultural sector on crops such as corn. As aldrin* is applied to soil and vegetation, residues will undoubtedly be present in the floral sector, atmosphere and soil complex shortly after application. Gannon and Bigger (33) have shown that aldrin is readily converted to dieldrin in plants and Thompson _et al._ (92) have studied the residue accumulation of pesticides in plant tissues such as peanut hay, shells and in the peanut itself. Dieldrin residues can be stored in plant tissue without ill effects providing the residues are not massive. The environmental ramifications would not be extremely complex if the residues were to remain in the floral sector or be broken down quickly (non-persistent). But, as with the majority of chlorinated hydrocarbon pesticides, dieldrin is highly resistant to biological and chemical modification. Therefore, dieldrin residues persist in the environment for long periods of time and are available for interaction with the remainder of the biosphere via various interface mechanisms. With this type of information one can begin to trace dieldrin's movement through the environment starting in the floral sector. Figure 12 indicates that the residue can be transferred directly to herbivores and man through ingestion and drinking, enter the soil complex (soil surface, soil sub-surface and soil organisms) via decay, and be translocated to the aquatic sector by runoff and erosion. The atmosphere acquires dieldrin residues by various means during the application process, primarily through particulate or vapor drift. Also, residues in soil can enter the atmosphere via wind erosion (abrasion). The atmosphere is in essence a transport mechanism in that there is little or no chemical or biological modification of pesticides while airborne. However, aldrin is epoxidized into dieldrin and both are photolyzed to photo-dieldrin (a much more highly toxic substance) in the air or on surfaces exposed to the UV component of sunlight. The distance over which a pesticide residue can be transported is dependent upon the compound's properties in accordance with physio-chemical principles. Removal of atmospheric residues is accomplished through washout by rain and dry deposition with the material entering the aquatic, soil and floral sectors. Therefore, the atmosphere may be seen as a cyclic system which absorbs, transports and deposits pesticide residues.

Soil Complex

When dieldrin is applied to corn, the soil complex is readily contaminated, with the soil surface receiving the greatest quantity of residues when compared to the soil sub-surface. Surface water containing residues may be leached (mostly in light sandy soils) to the groundwater and the soil sub-surface. The organic and inorganic particulates in both the surface and sub-surface soil complexes can accumulate dieldrin residues, with the organic matter having a greater affinity for the accumulation of chlorinated hydrocarbons.

---

*Aldrin is a soil-incorporated insecticide used to control lepidopterous and coleopterous larvae emerging from the soil to feed on the plants or feeding directly on the roots.

Within the soil complex, specifically the soil organisms, considerable concentration of dieldrin residues has been found to occur. Korschgen (52) found that earthworms *(Lumbricus terrestris)* accumulated 4.8 times the amount of dieldrin found in average soil and had a "storage ratio"* of 5.68. Korschgen (52) also found, as noted in the section on Insects p. 88, indications of biotransfer and biomagnification of dieldrin in three species of beetles. Gish (37) found that slugs usually have higher residues than earthworms from the same site. Thus far this evidence appears to suggest some food chain transfer from the residues in the soil (either from decaying plant or animal matter or from direct pesticide application to the soil) to the organisms inhabiting this part of the ecosystem. A note of caution must be observed, however. Since earthworms process soil, litter, microbiota and even minute macroinvertebrates, determination of the route of exposure and mechanism of accumulation is difficult.

The consequences of such residue transfer and magnification are of paramount importance to species' survival. Coaker (17) noted that sub-lethal residues of dieldrin caused extreme hyperactivity in carabid beetles *(Bembidion lampros)*. In conjunction with the alteration of individual activity patterns, sub-lethal residue accumulation can alter the predator-prey relationship. For example, after application of insecticides, the number of cabbage root flies increased, due to the demise of the root fly's natural enemies and the resultant lack of downward pressure on the cabbage root fly population. Sub-lethal residue accumulation can also result in the alteration of behavior patterns. For example, the prey (earthworm) may spend more time in an exposed position and may become more noticeable to the predator (birds). This results not only in the demise of large numbers of prey organisms but also in the consumption of greater numbers of contaminated prey organisms by the predator. This may allow accumulation of residues to a lethal dose in the predator or allow the higher concentration of residues to be passed on to a higher trophic level (49) (89).

Ingestion and consequent biotransfer represents only one mode of residue translocation from the soil complex. Runoff and erosion transport residues from the soil complex to the aquatic sector of the biosphere. Vegetation can also imbibe pesticide residues contained in the groundwater and wind erosion introduces residues into the atmosphere. This segment of the cycle is completed when residues return to the soil complex through decaying contaminated animal and vegetable matter, excreta deposition and rainfall and dry deposition.

Higher Terrestrial Trophic Levels

In the case of birds, Scott et al. (85) found that dieldrin applied to control insect pests in Illinois caused heavy mortalities in birds and mammals. Dustman and Stickel (24) noted that dieldrin used to control white-fringed beetles in Tennessee resulted in numerous bird and mammal deaths. Therefore, it is safe to surmise that dieldrin is deleterious to some species of birds and mammals. Birds that are carnivores (eating other birds and fish) usually have higher residues of dieldrin than those birds ingesting primarily seeds and vegetation, once again suggesting biotransfer of dieldrin residues from vegetation or soil to higher trophic levels. For more detailed analysis of residues in birds see Keith and Gruchy (51); Vermeer and Reynolds (94); Prestt, et al. (75). Studies done by Cummings et al. (13) (14) showed that dieldrin was readily stored in avian tissues, and Bailey et al. (2) and Friend (31) showed that dieldrin accumulated in the brain tissue of pigeons *(Columba livia)* and mallard ducks *(Anas platyrhynchos)*.

---

*Storage Ratio = $\frac{\text{ppm residues in earthworms}}{\text{ppm residues in soil}}$

First generatiun pheasants containing dieldrin had no abnormal physiological symptoms but second generation pheasants had significantly lower hatchability and fertility than controls (3). The chicks also had visual impairments. Shag *(Phalacrocorax aristotelis)* eggs contained 0.5-3.0 ppm of dieldrin suggesting the possible relationship between clutch viability and dieldrin residues since six of the eight clutches had no surviving chicks at the end of the tenth day (74). Dieldrin in sheep dip and residues in the environment seem to have caused poor reproduction in golden eagles in Scotland (58).

On still a higher level, large residues of dieldrin were found in the tissues of dead mammals in areas of the southeast U.S. treated with dieldrin for fire ant control. The affected organisms included cotton rats, white-footed mice *(Peromyscus sp.)*, cottontail rabbits *(Sylvilagus floridanus)*, raccoons *(Procyon lotor)*, armadillos *(Dasypus novemcinctus)* and red foxes *(Vulpes fulva)* (21). Similar findings occurred in Illinois when dieldrin was used to curb a Japanese beetle *(Popillia japonica)* infestation. Ground squirrels *(Citellus tridecemlineatus* and *C. franklinii)*, muskrats *(Ondatra zibethica)*, cottontail rabbits, short-tailed shrews *(Blarina brevicauda)*, fox squirrels *(Sciurus niger)*, woodchucks *(Marmota monax)* and prairie voles *(Microtus ochrogaster)* suffered heavy mortalities (85). Blackmore (4) reported increased mortality in British foxes *(Vulpes vulpes)* and it was suggested that the foxes had died from eating dead birds that had eaten grain treated with dieldrin. This incident serves to illustrate the importance of examining food chain relationships prior to pesticide-use decision in an attempt to forego serious environmental reactions. In this case dieldrin was transferred, in the form of residues, from the original application site (floral sector) up the food chain to birds and then to carnivores via ingestion. It is important to point out that movement of residues within the environment does not stop with the death of the foxes. Scavengers (e.g., omnivores) can ingest the contaminated carcasses, the residues can be returned to the soil reservoir via decay and the chemical residues can be transported to the acquatic sector through runoff, erosion, sedimentation and leaching from the terrestrial ecosystem.

The white-fringed beetle, fire ant, Japanese beetle and treated grain examples do not specifically refer to aldrin application on corn, but these incidents are germane to the evaluation. That is, since aldrin is employed quite extensively on corn, it is relatively safe to suspect that organisms in the application areas may be impacted similarly. It is important to note, however, that susceptibility to pesticides can vary greatly within organisms of the same species.

Residue accumulation may not only cause death directly, but may also produce chronic physiologic malfunctions. In a three-year study on the effect of dieldrin on the physiology and reproduction of white-tailed deer *(Odocoileus virginianus)*, Murphy and Korschgen (68) found that fawns were smaller at birth, there were greater post-partum mortality, the fawns had smaller thyroid and pituitary glands, and there was placental transfer of dieldrin.

Aquatic Sector

When aldrin is applied to corn, introduction of the pesticide contaminant into the aquatic ecosystem is primarily achieved through the surface runoff, aerial transport and direct application of pesticide to aquatic systems. The aquatic sector possesses massive sorptive capacity and acts as a pollutant sink yielding a continual source of residues (90). Once the residues are in the aquatic sector, organisms can acquire contaminants by direct uptake of water-borne residues or indirectly through the ingestion of contaminated food stuffs, sediment and detrial material.

The uptake of dieldrin from food and water by fish and fish-food organisms has been studied by many including, Gakstatter and Weiss (32), Reinert (79), Hansen (40), and Holden (43). Some species can accumulate residue concentrations that are several thousand times greater than those in the water.

Petrocelli and Anderson (71) designed a study to test the bio-magnification of dieldrin residues in the food chain of clams and blue crabs. Analysis of the data found that animals fed small amounts of dieldrin at relatively low rates over short periods of time accumulated dieldrin up to 6.8 times the daily exposure. For actual experimental design and results see Petrocelli and Anderson (71). These results imply that under natural conditions, animals consuming contaminated food for months or years can accumulate high residue concentrations.

In another research effort by Petrocelli, Hanks and Anderson (72), the uptake and accumulation of dieldrin by esturine mollusck, *Rangia cuneata*, was examined. It was found that the mollusc accumulated dieldrin residues from extremely low levels in the water.

It is imperative that the fate of chemical agents in the ecosystem be established prior to a pesticide's wide-scale use. Metcalf et al. (65) developed a model ecosystem in which the movement of a pesticide and its metabolites could be examined. Using the Metcalf model, Sanborn and Yu (83) added one crab *(Uca manilensis)* and a water plant, *Elodea*, to the original model and traced the movement of dieldrin. The fish at the top of the food chain in the model concentrated dieldrin approximately 6,000 times the concentration in the water. *Physa* snails accumulated 115,000 times the concentration in the water (about nineteen times the value found in fish). It was found that dieldrin is highly resistant to chemical or biological modification which, when related to the real environmental case, indicates that dieldrin residues from any given application will remain in the environment for some time to come.

Hannon (39) observed that fish at high tropic levels have large concentrations of residues, one of which is dieldrin. Exposure to these sub-lethal concentrations and the subsequent accumulation of chemicals by fish can result in physiologic malfunctions having far-reaching environmental ramifications. Research done by Burdick et al. (6) indicates that exposure to sub-lethal concentrations of dieldrin may result in higher mortality through increased reproductive failures (gamete residues) and greater susceptibility to predation due to alteration of fry mobility and activity patterns. In addition, dieldrin exposure reduced both growth and fecundity in *Poecilia latpinna* (54). Cain (8) found that exposure to dieldrin reduced the resistance to disease and produced below normal feeding rates. Johnson (50) observed thickening of gill membranes, lack of osmoregulation, lower blood counts, brain damage, and reduced body weight in fish exposed to dieldrin.

A suitable example of the movement of dieldrin in the marine food chain has been done by Robinson and is illustrated in Figure 14. This food chain model shows the biotransfer and biomagnification potential of dieldrin within this section of the ecosystem. It is by a composite of models such as shown in Figure 14 that an evaluator may identify a pesticide's possible movement through the environment.

Summary

Since dieldrin is resistant to chemical and biological modification, it can and does persist in all facets of the environment. This persistence affords the opportunity for biotransfer and biomagnification of residues in the biosphere as exemplified in the test. Food chain models such as Figure 17 will afford greater comprehensiveness of analysis and consequently facilitate a comprehensive view of environmental interdependencies and pesticide impacts.

With the application of aldrin to corn, there exists a myriad of possibilities for environmental contamination. The aforementioned effects of aldrin aid in the illustration of the flow of a pesticide through the food chain model, but the models are operational even if little or nothing is known about actual *in vivo* consequences.* Based on laboratory experimentation, the inherent properties of the pesticide in question, and knowledge of food chain relationships, the effects of a pesticide can be foreseen, to some extent facilitating the minimization of ecological "surprises."

Once the potential impact areas are identified, there still remains the question of how to evaluate these environmental impacts and integrate them into a benefit-cost framework. In tracing the movement and subsequent environmental effects of aldrin applied to corn, the impacted organisms are entered into the environmental impact matrix where an attempt at quantification can commence.

---

*Unfortunately, little is known about environmentally unstable compounds that do not leave residues but, which, by virtue of repeated applications or some sustained release mechanism, provide iterative or chronic exposure.

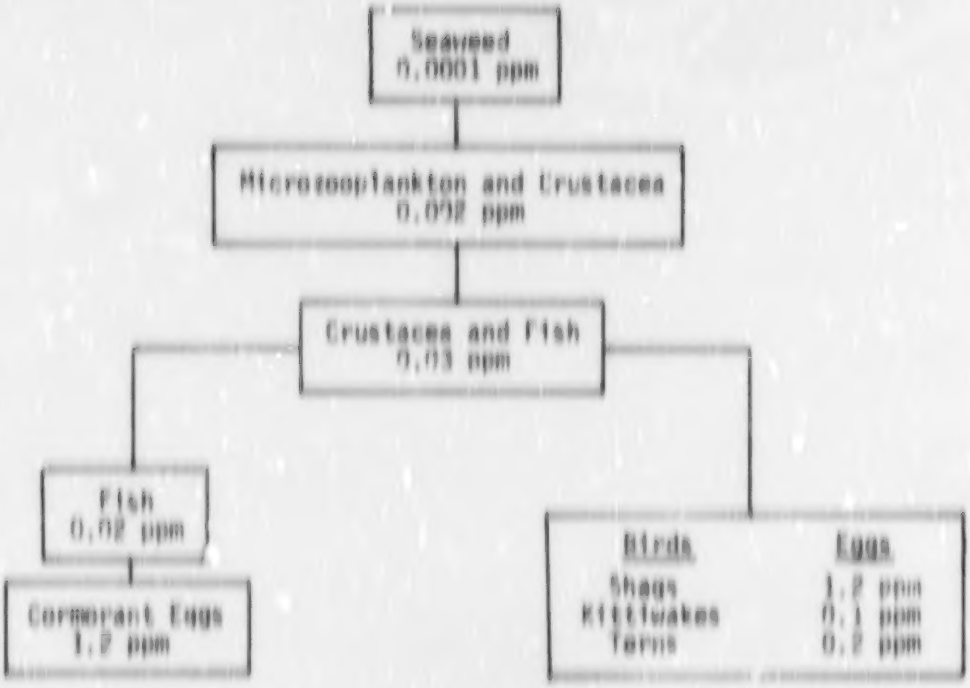

Seaweed
0.0001 ppm
Microzooplankton and Crustacea
0.032 ppm
Crustacea and Fish
0.03 ppm
Fish
0.02 ppm
Cormorant Eggs
1.7 ppm
Birds
Eggs
Shags
1.2 ppm
Kittiwakes
0.1 ppm
Terns
0.2 ppm

Figure 14. Concentration of dieldrin in marine organisms [Robinson, [82]].

ALDRIN-DIELDRIN IMPACT MATRIX

Matrix Operation

Using the environmental impacts delineated earlier in this section an impact matrix such as that shown in Figure 15 can be developed. The abscissa consists of those species (both flora and fauna) which are impacted and the ordinate is comprised of aldrin-dieldrin induced effects. A slash is placed at the intersection of the impacted organism and the effect. This process is continued until the evaluator has entered all the impacts that he is aware of based on his knowledge of the relevant literature, consultation with colleagues, etc. Evaluation of the impacts can then commence. A number (ranging from a high of ten to a low of one) in the upper left-hand corner indicates the magnitude or extent of the impact on the organisms under consideration while the number (high of ten to low of one) in the lower right-hand corner is a relative importance rating of the effect. The importance rating is based on the organism's place and function in the food chain as well as its intrinsic or market value to man.

For example, earthworms readily accumulate dieldrin residues which are retained in the organism affording greater possibilities for food chain transfer [illegible] residues. It is important to note that uptake of residues by organ[illegible] not imply storage and accumulation. But, in the case of the earthworm, the uptake and accumulation warrants a magnitude rating of five.* Since the dieldrin resource can alter behavior patterns making the earthworm more noticeable and therefore easier prey, the magnitude and importance of behavior alterations is assessed at four and eight, respectively. Earthworms are a food source for many birds and some mammals which also tend to accumulate dieldrin residues quite readily thus reinforcing the residue accumulation importance rating of nine for earthworms. It should be noted that as higher trophic levels are reached, the importance rating shifts from strictly food chain criteria to one based on man's valuation of the impacted organisms. This is illustrated with the case of the golden eagle. It is at the top of its food chain and has a low importance rating in that sense but is assumed to be highly valued by man since it is protected by law. From this perspective, the importance of the impact is high, assessed at ten.

The significance of food chain relationships to impact assessment can readily be seen since the importance rating of residue accumulation in a species is a function of its level and importance in the food web hierarchy. In tracing the impact assessment of an organism through the matrix, keeping in mind the underlying food chain interdependencies, a comprehensive picture of pesticide impacts can be developed and then evaluated. This concept is

---

*All numerical ratings in this section were developed by the author for illustrative purposes only and do not necessarily reflect the judgment of experts in biological effects of chemical pesticides as would be the case in an actual assessment for registration decision purposes.

Reproduced from
best available copy

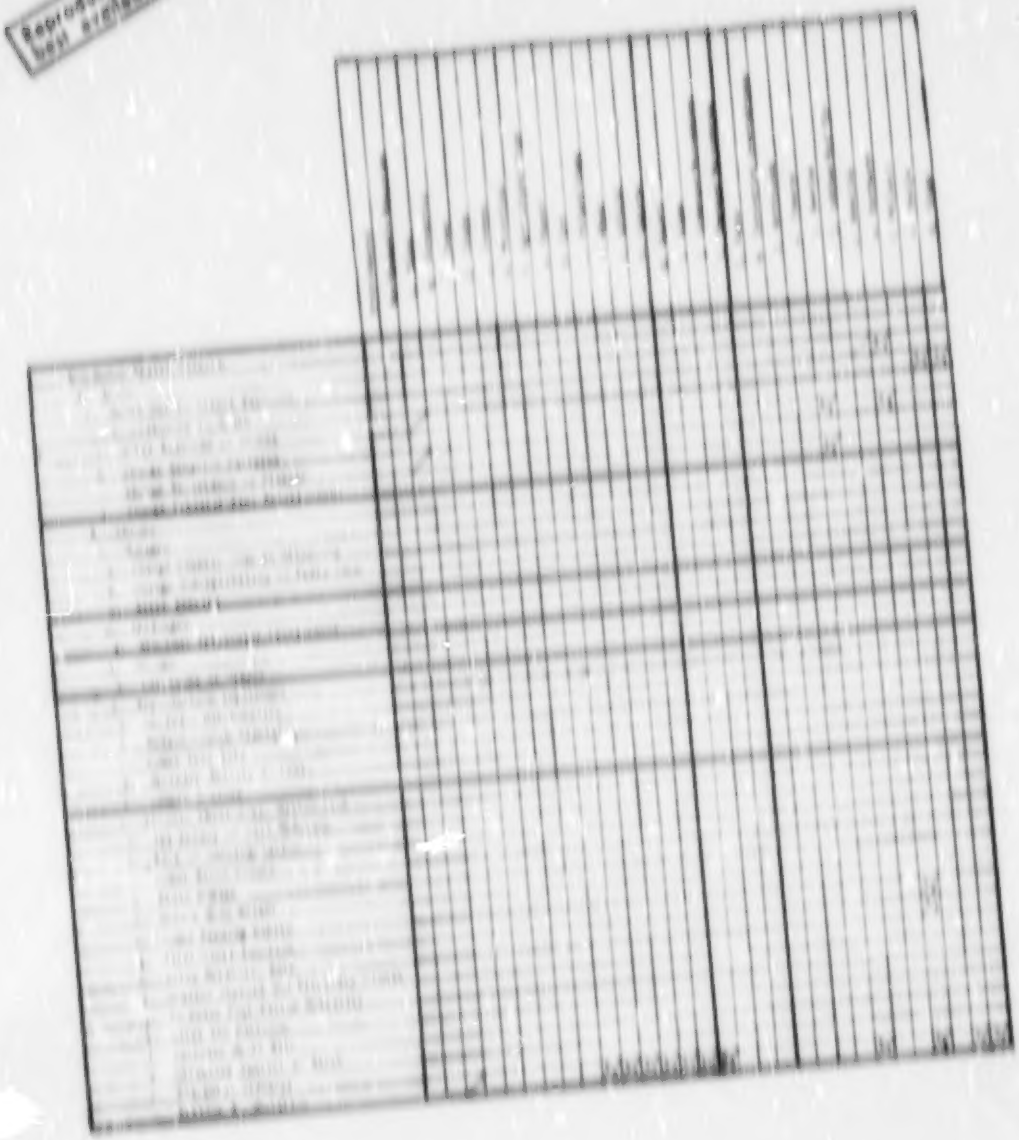


Figure 15. Aldrin-Dieldrin impact matrix.

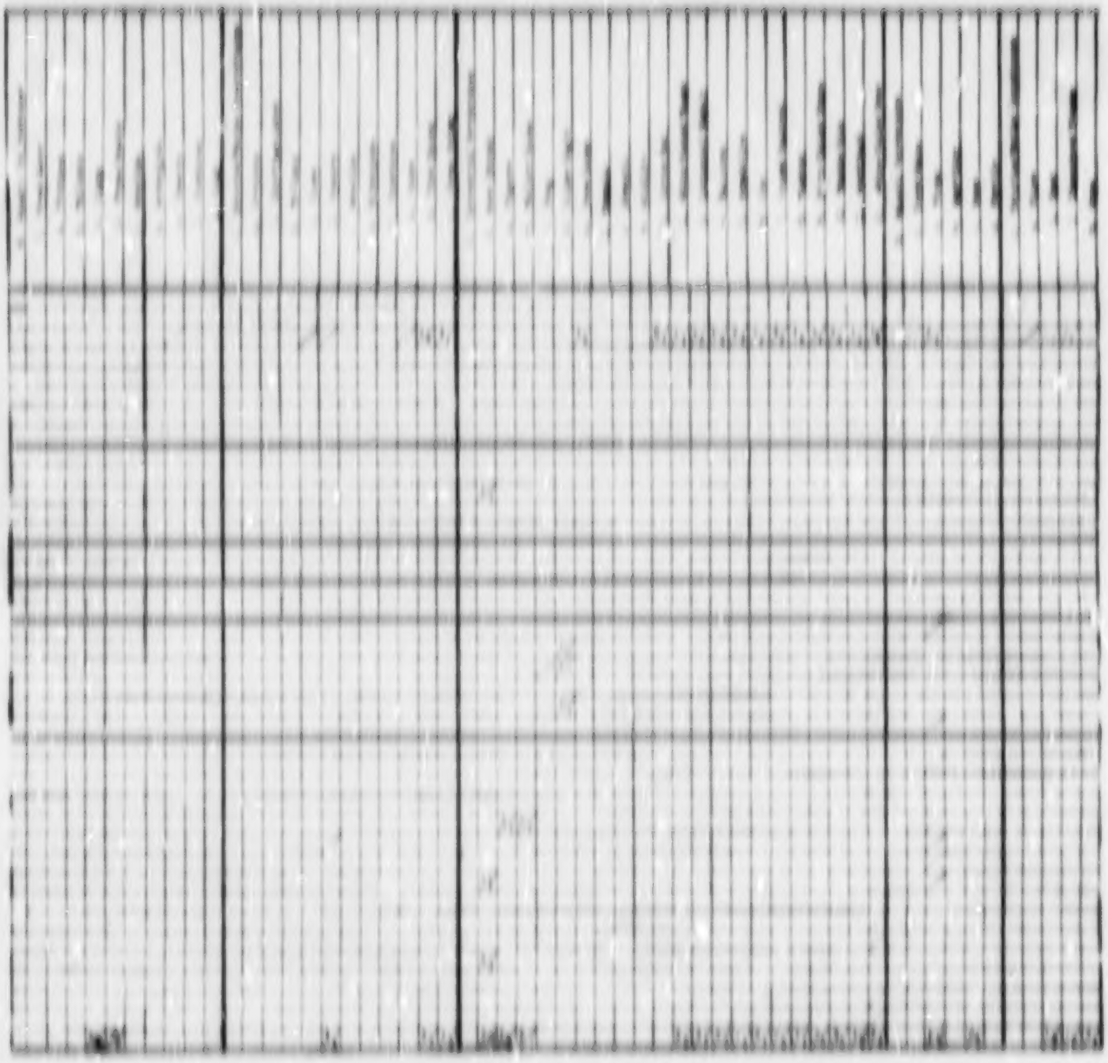


Figure 15 (continued)

important since a pesticide may strongly impact a particular segment of the environment but the importance of this effect on the entire ecological scheme may be small.

An example will help to clarify this point. Snails and slugs are adversely impacted by dieldrin and accumulate residues to approximately the same extent as earthworms. However, the accumulation of residues in snails and slugs is not as important as earthworms since snails and slugs are not as readily preyed upon as earthworms. This is most likely due to the slimy mucus coating of the soil molluscs to which potential predators display aversion. Without predation from higher trophic levels, the residue-laden snails and slugs do not present a serious environmental threat by way of food chain transfer. Therefore, a residue-accumulation importance rating of five seems appropriate.

It may prove advantageous to trace the impact assessment through the matrix to provide a holistic view of the integrated food chain model and assessment matrix in operation. When aldrin is supplied to corn for soil insect control, residues (dieldrin) become present in the soil and are available to non-target organisms both directly and indirectly. Soil organisms (e.g., earthworms) can accumulate the dieldrin residue by ingestion and absorbtion yielding an impact rating of four and eight (magnitude and importance, respectively). Since dieldrin residues can alter behavior in the earthworm, and it is a food source for many carnivores (both birds and mammalian insectivores), the residue impact is assessed at five and nine. Small mammals, such as shrews, voles, etc., accumulate residues from ingesting contaminated food stuffs (earthworm) and in turn can be killed directly due to dieldrin toxicity or can accumulate residues themselves. Once again, the importance of residue accumulation in these organisms is embodied in the animals' food web integration. Since these small mammals are consumed by predators (both birds and mammals), residue accumulation warrants an importance assessment of eight. If the golden eagle ingests these contaminated organisms, the magnitude of the impact (reproductive failure) may be high due to a several-fold increase in the proportion of birds that do not reproduce successfully. The importance rating (10) is also quite high due in part to species scarcity and man's valuation of this somewhat rare bird. The significance of the golden eagle as a food source is minimized by virtue of its standing near the top of the predator-prey hierarchy.

The importance rating on organisms that exhibit acute toxicity to aldrin and dieldrin will depend primarily on species abundance and specialized food chain relationships. Therefore, the importance rating of an organism's death may substantially differ from the importance evaluation of that organism's residue accumulation potential. Critical to the importance rating is the definition of perspective. The majority of pesticide expertise is centered around protection/production with this particular illustrative effort taking the latter perspective in that protection is applied to biological systems. The inherent bias in these stances could possibly be counter balanced by the selection of experts in the field employed by non-production groups.

Clearly, the food chain relationship presented is only one of many possibilities which could be presented, but the example suffices to illustrate the basic rationale underlying the impact assessment scheme. Many of the impact ratings in Figure 15 have not been discussed, but the assessment of effects follows the same strategy set forth in the example.

Figure 15 represents the efforts of only one evaluator and it is recognized that there may be numerous other entries possible. For this reason, the Delphi technique and an "open-ended" matrix are proposed to insure that the evaluation of possible aldrin-dieldrin induced impacts is comprehensive. The aggregation of many responses to a matrix such as Figure 15 will yield a composite impact matrix which decision-makers may use in evaluating a particular pesticide. The output from the environmental assessment, when integrated with the information obtained from production, human health, aesthetic, and distribution analysis, will afford the decision-maker vital data for benefit-cost analysis of pesticide-use decisions.

Summary

The majority of aldrin (which degrades to dieldrin) is used by the agricultural sector on corn. The persistance of aldrin is agronomically desirable, but this characteristic also has serious environmental ramifications. The long life of aldrin permits translocation from the application site to other sectors of the environment. This chemical stability also facilitates the contamination of non-target organisms, possibly man himself. The examples presented in Section 15 indicate the types of effects that aldrin-dieldrin induce in the environment. This *ex ante* view of environmental impacts, however, is not the most desirable method by which to evaluate aldrin-dieldrin effects since impacts can be of an irreversible nature. The systematic examination and evaluation of impacts facilitated by the food chain models and measurement techniques developed in this research effort conclude that the environmental impacts of aldrin-dieldrin are extensive. And, since this is in essence only a partial analysis, it is reasonable to suggest that the ecological consequences induced by this particular pesticide could be even more widespread.

Man only becomes aware of environmental alteration at a relatively high trophic level. The acute toxicity of aldrin is of great biotic importance, but is subordinate to the chronic effects of dieldrin. Through residue accumulation and biomagnification, dieldrin moves from the application point and low trophic levels to non-target area and higher trophic levels, possibly to man himself. This environmental evaluation scheme begins with a microsystem stance in an attempt to predict macrosystem consequences. The material presented illustrates that the ecological effects of aldrin-dieldrin are extensive in both a micro and macro system framework while, at the same time, emphasizing environmental interdependencies. Residue accumulation and biomagnification of dieldrin, as related to man's perception of ecological alteration, is by far the most important effect.

As previously discussed, incorporation of environmental data into a benefit-cost or other analytical frame works is imperative. The most desirable state, when using benefit-cost analysis, would be to reduce environmental impacts to monetary terms making the units commensurate with other variables in the analysis. Unfortunately, this conversion can only be accomplished in part and under highly specialized analytical conditions. The expenditures approaches used by hunting and fishing agencies are not adequate vehicles for environmental cost estimation in that the schemes possess no capability for species specificity and yield defective measures of aggregate wildlife value. If environmental parameters cannot be totally and correctly reduced to monetary terms, it would be inadvisable to incorporate these values into a benefit-cost structure. Rather than using conceptually erroneous measures of environmental impacts, we recommend the evaluation scheme set forth in this section and comparison of environmental consequences among alternative pesticide-use decisions.

BIBLIOGRAPHY FOR ENVIRONMENTAL IMPACTS CHAPTER

1. Appert, J. Docum. Inst. Rech. Agron. Madagascar, 113, 1967.

2. Bailey, S., J. Bunyan, B. D. Rennison and A. Taylor. The Metabolism of 1,1-Di (p-chlorophenyl)-2,2-dichloroethylene and 1,1-Di (p-chlorophenyl)-2-chloroethylene in the Pigeon. Toxicol. Appl. Pharmacol., 14: 23-32, 1969.

3. Baxter, W. L., R. L. Liner and R. B. Dahlgren. Dieldrin Effects on Two Generations of Penned Hen Pheasants. J. Wildl. Mgmt., 33(1): 96-103, 1969.

4. Blackmore, D. K. The Toxicity of Some Chlorinated Hydrocarbon Insecticides to British Wild Foxes *(Vulpes vulpes)*. J. Comp. Pathol. and Therap., 73: 391-409, 1963.

5. Boykins, E. A. DDT Residues in the Food Chain's of Birds. Atlantic Nat., 21: 18-25, 1966.

6. Burdick, G. E., E. J. Harris, H. J. Dean, T. M. Walker, J. Shea and D. Colby. The Accumulation of DDT in Lake Trout and the Effect on Reproduction. Trans. Am. Fish. Soc., 93: 127, 1964.

7. Butler, L. I., B. J. Landis and L. M. McDonough. Circular No. 522, Washington Agr. Exp. State., College of Agr., Washington State University, 1970.

8. Cain, S. A. Pesticides in the Environment with Special Attention to Aquatic Biology Resources. Rept. U.S. Japan Meeting on Pesticide Research, 1965.

9. Cicchetti, Charles J. and Myrick Freeman. Option Demand and Consumer Surplus: Further Comment. Q.J.E., 85(3): 538, August 1971.

10. Coaker, T. H. The Effects of Soil Insects on the Predators and Parasites of the Cabbage Root Fly *(Erioischia brassicae (Bouche))* and on the Subsequent Damage Caused by the Pest. Ann. Appl. Biol., 57: 397-407, 1966.

11. Collingwood, C. A. Persistant Pesticides - A Look At Future Needs. Ann. Appl. Biol., 56: 336-337, 1965.

12. Conney, A. H., R. M. Welsh, R. Kuntzman and J. J. Burns. Clin. Pharmacol. Ther., 8: 2, 1967.

13. Cummings, J. G., M. Eidelman, V. Turner, D. Reed, K. T. Zee and R. E. Cook. Residues in Poultry Tissues from Low Level Feeding of Five Chlorinated Hydrocarbon Insecticides to Hens. J. Assn. Offic. Anal. Chem., 50: 418, 1967.

14. Cummings, J. G., K. T. Zee, V. Turner and F. Quinn. J. Ass. Offic. Anal. Chem., 49: 354, 1966.

15. Dalkey, N. C. The Delphi Method: An Experimental Study of Group Opinion. RAND Corp., RM-5888-PR, 1969.

16. Dalkey, N. C. Delphi. the RAND Corp., P-3704, October, 1967.

17. Dalkey, Norman C. and Daniel L. Rourke. The Delphi Procedure and Rating Quality of Life Factors. In The Quality of Life Concept: A Potential New Tool for Decision Makers. EPA, Office of Research and Monitoring, Environment Studies Division, 1973.

18. Davis, B. N. K. Soil Animals as Vectors of Organochlorine Insecticides for Ground-Feeding Birds. J. Appl. Ecol., 3: 133-139, 1966.

19. Dee, Norbert, *et al.* Environmental Evaluation System for Water Resource Planning. Report to the U.S. Bureau of Reclamation, Columbus, Ohio. Battelle Memorial Institute, Jan., 1972.

20. Dee, Norbert, *et al.* Planning Methodology for Water Quality Management: Environment Evaluation System. Columbus, Ohio, July, 1973.

21. Dewitt, J. B., C. M. Menzie, V. A. Adomaitis and W. L. Reichel. Pesticidal Residues in Animal Tissues. Trans. 25th N. Amer. Wildl. Conf. 227, 1960.

22. Dimond, J. B. and J. A. Sherburne. Persistance of DDT in Wild Populations of Small Mammals. Nature (London), 22(5179) 486-487, 1969.

23. Dorough, H. W. and N. M. Randolph. Comparative Residual Nature of Certain Insecticides Applied as Low Volume Concentrate and Water Emulsion Sprays. Bull. Environ. Contam. and Toxicol., 2(6): 340-348, 1967.

24. Dustman, E. H. and L. F. Stickel. Pesticides and Their Effect on Soil and Water. Amer. Soc. Agron. Spec. Publ. No. 8. Soil Sci. Soc. Amer., 1966, pp. 109-121.

25. Edwards, C. A., ed. Environmental Pollution by Pesticides. Plenum Press, London and New York, 1973.

26. Edwards, C. A. Persistant Pesticides in the Environment. CRC Press, Cleveland, Ohio, 1970.

27. Edwards, C. A. Critical Reviews in Environmental Control. Vol. 1, Issue 1, Chem. Rubber Company, 1970, pp. 7-76.

28. Edwards, C. A., J. R. Lofty, A. E. Whiting and K. A. Jeffs. Rep. Rothanest. Exp. Sta. for 1970, Vol. 193, 1970.

29. Ferguson, D. E. Trans. 32nd N. Amer. Wildl. and Natural Resources Conf., 1967, pp. 103-107.

30. Ferguson, D. E. In Reservoir Fisheries Resources Symp. Reservoir Committee on the So. Div. Amer. Fish Soc., University of Georgia, Athens, 1968, pp. 531-536.

31. Friend, M. Pesticide-Infectious Disease Interaction Studies in the Mallard. Ph.D. Thesis, University of Wisconsin, 1971.

32. Gakstatter, J. H. and C. M. Weiss. The Elimination of DDT-C^{14}, Dieldrin-C^{14}, and Lindane-C^{14} from Fish Following A Single Sublethal Exposure in Aquaria. Trans. Am. Fish. Soc., 96(1): 301-307, 1967.

33. Gannon, N. and J. H. Bigger. The Conversion of Adrin and Heptachlor to Their Epoxides in Soil. J. Econ. Entomol., 51(1): 8-11, 1958.

34. Gillett, James W. Personal communication, 1976.

35. Gillett, James W., James Hill IV., Alfred W. Jarvinen, and W. Peter Schoor. A Conceptual Model for the Movment of Pesticides Through the Environment. National Ecological Research Lab., Gulf Breeze Environmental Research Lab., National Water Quality Lab., Southeast Environmental Research Lab., National Environmental Research Center, Corvallis, Oregon, Dec. 1974.

36. Gillet, James W., ed. The Biological Impact of Pesticides in the Environment. Proceedings of the Symposium, August 18-20, 1969, Oregon State University, Corvallis, Oregon, 1970.

37. Gish, C. D., Pestic. Monit. J., 3(241), 1970.

38. Goldberg, E. D., P. Butler, P. Meier, D. Menzel, C. Paulik, R. Risebrough and L. F. Stickel. National Academy of Sciences, Washington, D.C., 1971.

39. Hannon, M. R. M. S. Thesis, South Dakota State Univ., 1966.

40. Hansen, D. J. An Annual Report of the Bureau of Commercial Fisheries Biological Lab. Gulf Breeze, Fla., for the Fiscal Year Ending June 30, 1965. Bureau of Commercial Fisheries Circ. No. 247, 10, 1966.

41. Harris, C. R. Proc. Entomol. Soc. Ontario, 100: 14, 1969.

42. Herman, S. G., R. L. Garret and R. L. Rudd. Chemical Fallout. C. C. Thomas, Springfield, Ill., 1969, pp. 24-51.

43. Holden, A. V. Organochlorine Insecticide Residues in Salmonid Fish. J. Appl. Ecol., 3:45-53, 1966.

44. Howe, C. W. Benefit-Cost Analysis for Water System Planning. Water Resource Monograph 2, American Geophysical Union, 1971.

45. Hunt, L. B. Kinetics of Pesticide Poisoning in Dutch Elm Disease Control - The Effects of Pesticides on Fish and Wildlife. U.S.D.I. Fish and Wildl. Serv. Circ. 226. Washington, D.C. 12, 1965.

46. Hunt, L. B. and R. J. Sacho. Response of Robins to DDT and Methoxychlor. J. Wildl. Mgmt., 33(2): 336-345, 1969.

47. Ingram, W. R. Cott. Grow. Rev., 44: 203, 1967.

48. Jefferies, D. J. Meded. Fakult, Landbouw. Gent., 36: 34, 1971.

49. Jefferies, D. J. and C. H. Walker. Uptake of pp'-DDT and Its Post-mortem Breakdown in the Avian Liver. Nature (London), 212: 533-534, 1966.

50. Johnson, D. W. Pesticides and Fishes - A Review of Selected Literature. Trans. Amer. Fish. Soc., 97: 398, 1968.

51. Keith, J. A. and I. M. Gruchy. Proc. 15th Int. Ornith. Congress, 1970.

52. Korschgen, L. J. Soil - Food Chain - Pesticide Wildlife Relationships in Aldrin Treated Fields. J. Wildl. Mgmt., 34(1): 186-199, 1970.

53. Kulash, W. M. Soil Treatment for Wireworms and Cutworms. J. Econ. Entomol., 40(6): 851-854, 1947.

54. Lane, C. E. and R. J. Livingston. Some Acute and Chronic Effects of Dieldrin on Sailfin Molly, *Poecilia latipinna*. Trans. Am. Fish. Soc., 99(3): 489-495, 1970.

55. Lazarevic, B. M. Effects of the Number of Sprayings and Different Chemicals in Controlling Cotton Pests in the Sudan. J. Econ. Entomol., 63(2): 629-633, 1970.

56. Leopold, L. B., et. al. A Procedure for Evaluating Environmental Impact. Geological Survey Circular 645, U.S. Geological Survey, Washington, D.C., 1971.

57. Little, A. D., Inc. Transportation and Environment: Synthesis for Action: Impact of National Environmental Policy Act of 1969 on the Department of Transportation, Vol. 3, Options for Environmental Management, prepared for Office of the Secretary, Department of Transportation, July, 1971.

58. Lockie, J. D., D. A. Ratcliffe and R. Balharry. Breeding Success and Organochlorine Residues in Golden Eagles in West Scotland. J. Appl. Ecol., 6: 381-389, 1969.

59. Lockie, L. N. and D. A. Ratcliffe. Insecticide and Scottish Golden Eagles. Brit. Birds, 57: 89-102, 1965.

60. Macek, J., Environmental Health Sciences Series No. 1, Oregon State University, Corvallis, Oregon, 1970.

61. Materu, M. E. A. and D. Hopkins. E. Afr. Agric. For J., 35(70), 1969.

62. McHarg, Ian. A Comprehensive Highway Route-Selection Method. Highway Research Record, Number 246, 1968, pp. 1015, and McHarg, Design With Nature, Natural History Press, Garden City, New York: 1969, pp. 31-41.

63. Mehner, J. F. and G. J. Wallace. Atlant. Nat., 14: 4, 1959.

64. Metcalf, R. L. Proc. 2nd Internat. Plant. Prot. Conf., Vol. 129, 1967.

65. Metcalf, R. L., I. P. Kapoor and S. K. Gurchavan. Model Ecosystem for the Evaluation of Pesticide Biodegradability and Ecological Magnification. Environ. Sci. Tech., 5(8): 709-713, 1971.

66. Moriarty, F. The Toxicity and Sublethal Effects of p, p'-DDT and Dieldrin to *Aglais urticae* (L.) (Lepidoptera: Nymphalidae) and *Chorthippus brunneus* (Thunberg) (Saltatoria: Acrididae). Ann. Appl. Biol., 62: 371-393, 1968.

67. Murphy, P. G. The Effect of Size on the Uptake of DDT from Water by Fish. Bull. Environ. Contam. and Toxicol., 6(1): 20-23, 1971.

68. Murphy, D. A. and L. J. Korschgen. Reproduction, Growth, and Tissue Residue of Deer Fed Dieldrin. J. Wildl. Mgmt., 34(4): 887-903, 1970.

69. Ogilvie, D. M. and J. M. Anderson. J. Fish. Res. Bd. Canada, 22: 503, 1965.

70. Peterle, T. J. In The Biological Impact of Pesticides in the Environment. J. W. Gillett, ed., Environmental Health Sciences Center, Oregon State University, Corvallis, Oregon, 1970.

71. Petrocelli, S. R. and J. Anderson. Biomagnification of Dieldrin Residues by Food-Chain Transfer from Clams to Blue Crabs Under Controlled Conditions. Bull. Environ. Contam. and Toxicol., 13(1): 108-122, 1975.

72. Petrocelli, S. R., A. R. Hanks and J. Anderson. Uptake and Accumulation of an Organochlorine Insecticide (Dieldrin) by an Esturine Mollusc, *Rangia cuneata*. Bull. Environ. Contam. and Toxicol., 10(5): 315-320, 1973.

73. Pill, J. The Delchi Method: Substance, Context, A Critique and Annotated Bibliography. Socio-Econ. Plan. Sci., 5(1), 1969.

74. Potts, G. R. Success of Eggs of the Shag on the Farne Islands, Northumberland, in Relation to Their Content of Dieldrin and pp'-DDE. Nature (London), 217: 1282-1284, 1968.

75. Prestt, I., D. J. Jefferies and N. W. Moore. Polychlorinated Biphenyls in British Wild Birds. Env. Poll., 1(1): 3-27, 1970.

76. Pritchard, H. N. and D. M. Dines. DDT Effects on Oxygen Evolution During Photosynthesis. Proc. Pa. Acad. Sci., Vol. 46, 1972, pp. 25-26.

77. Ratcliffe, D. A. Changes Attributable to Pesticides in Egg Breakage Frequency and Eggshell Thickness in Some British Birds. J. Appl. Ecol., 7: 67-107, 1970.

78. Reed, W. Use of Insecticides on Cotton in Western Tanzania. Agric. Vet. Chem., 8(1): 8-10, 1967.

79. Reinert, R. E. The Accumulation of Dieldrin in an Alga *(Scenedesmus obliquus)*, Daphnia *(Daphnia magna)*, Guppy *(Lebistes reticulatus)* Food Chain. Ph.D. Thesis, Univ. of Michigan, Ann Arbor, Michigan, 1967, pp. 76.

80. Ridgeway, R. L., D. A. Lindquist and D. L. Bull. Effect of Method of Application on Uptake of Di-Syston by the Cotton Plant. J. Econ. Entomol., 58(2): 349-352, 1965.

81. Robertson, I. A. D. E. Afr. Agric. For. Ind. J., 35: 181, 1969.

82. Robinson, J. Organochlorine Insecticides and Birds. Chem. Brit., 4: 4, 1968.

83. Sanborn, J. R. and C. C. Yu. The Fate of Dieldrin in a Model Ecosystem. Bull. Environ. Contam. and Toxicol., 10(6): 340-346, 1973.

84. Satchell, J. E., A. Burges and F. Raw, ed. Soil Biology, Academic Press, London and New York, 1967.

85. Scott, T. G., Y. L. Willis and J. A. Ellis. Some Effects of A Field Application of Dieldrin on Wildlife. J. Wildl. Magmt., 23(4): 409-427, 1959.

86. Skutch, M. and J. L. Schofer. Goals - Delphis for Urban Planning: Concept in Their Design. Socio-Econ. Plan. Sci., 7(3), 1973.

87. Smith, W. I. Quantifying the Environmental Impact of Transportation Systems. Van Doren-Hazard-Stallings-Schnacks, Topeka, Kansas, undated.

88. Stickel, W. H., L. F. Stickel and J. W. Spann. N. W. Miller and G. G. Berg, ed. Chemical Fallout. C. C. Thomas, Springfield, Ill., 1969, pp. 174-204.

89. Stickel, W. H., D. W. Hayne and L. F. Stickel. Effects of Heptachlor-Contaminated Earthworms on Woodcocks. J. Wildl. Mgmt., 29: 132-146, 1965.

90. Stout, V. F. Pesticide Levels in Fish of the Northeast Pacific. Bull. Environ. Contam. and Toxicol., 3(4): 240-246, 1968.

91. Stringer, A. and J. A. Pickard. The DDT Content of Soil and Earthworms in an Apple Orchard at Long Ashton. Rep. Agric. Hort. Res. Stn. Univ. Bristol for 1963. p. 127-131, 1965.

92. Thompson, N. P., W. B. Wheeler and A. J. Norden. Residues in Three Peanut Varieties Grown in Dieldrin Treated Soils. J. Agri. Food Chem., 18(5): 862-863, 1970.

93. USDA, Agricultural Research Service and Office of Research and Development, EPA. Control of Water Pollution From Cropland, Volume I - A Manual for Guildline Development. Revised Draft, April 1975.

94. Vermeer, K. and L. M. Reynolds. Organochlorine Residues in Aquatic Birds in the Canadian Prairie Provinces. Can. Field-Naturalist, 84: 117-130, 1970.

95. Warner, M. L. and E. H. Preston. Review of Environmental Impact Assessment Methodologies. U.S. Environmental Protection Agency, April 1974.

96. Westlake, W. E. and F. A. Gunther. Organic Pesticides in the Environment. R. F. Gould, ed. Advances in Chemistry Series 60, American Chemical Society, Washington, D.C., 1966, p. 110.

97. Wheatley, G. A. The Assessment and Persistence of Residues of Organochlorine Insecticides in Soils and Their Uptake by Crops. Ann. Appl. Biol., 55:325-329, 1965.

98. Wheeler, W. B., D. E. Frear, R. O. Mumma, R. H. Hamilton and R. C. Cotner. Absorption and Translocation of Dieldrin by Forage Crops. J. Agr. Food Chem., 15(2): 231-234, 1967.

99. Wright, B. S. Some Effects of Heptachlor and Dieldrin on New Brunswick Woodcocks. J. Wildl. Mgmt., 29(1): 172-184, 1965.

CHAPTER 6

AESTHETIC EFFECTS[*]

Closely related to the environmental effects is society's perception of its surroundings -- what we have chosen to call aesthetic effects. The close relationship can be seen when one considers that the natural environment plays a large part in one's perception of the quality of life. Even the urban resident can be affected if he realizes that the birds which were once in the park or the trees which were once on the corner are no longer there. In fact, the urban resident may be the most avid pursuer of the natural environment for its aesthetic component since research has shown that "individuals tend to seek elements which are lacking in their non-leisure environment" (6, p. 132).

Pesticide use can materially affect society's perception of its surroundings. The home use of pesticides is almost entirely devoted to increasing or at least maintaining an aesthetic level. Perhaps the only home uses of pesticides which have major non-aesthetic components are the use of pesticides on home vegetable gardens, and for home rodent control, termite control and anti-fungal treatments. Even in these cases, the aesthetic element cannot be dismissed. The use of pesticides by public agencies sometimes have major aesthetic components. Spraying for control of gypsy moth in the Northeast can be included here. The denuding of whole forests by insects can have a major effect on those viewing the scene as well as producing massive ecological disruptions. The aesthetic value of the view has been decreased and spraying the area, killing the insects and allowing the forest to recover may restore the value which had been lost. There obviously are very real aesthetic benefits which can stem from the use of pesticides. The use of pesticides for aesthetic improvement is not, however, the major problem.

As previously discussed under environmental effects, pesticide residues can have a major impact on various species of animals and plants. When this impact is large enough, there may be a perceived lowering in the aesthetic component. A well-known example of this is the robin poisonings due to their consumption of earthworms which had fed on elm leaves sprayed with a pesticide for control of Dutch elm disease. Judging from the response to the poisoning, one would say that robins were an important part of the surroundings of that city. The social cost of the spraying seemed to be high. Can this component of pesticide use be measured?

---

[*] Principal researcher in this section is Mr. Gary A. Shute, Center for the Study of Environmental Policy, The Pennsylvania State University. This section was prepared under the supervision of Dr. Donald J. Epp, Principal Investigator.

Some progress has been made in the past ten years in measuring social costs of externalities. The focus has been on air and water pollution, however, and it was hoped that the measurement techniques developed for these externalities could be adapted to pesticide residues and their externalities. Such an adaptation has proved impossible.

PROBLEMS OF QUANTIFICATION

Negative pesticide-related aesthetic affects are different from the negative aesthetic affects attributable to water and air pollution. This is true even though pesticide residues are a part of the total water and air pollution problem. This can be seen more easily if the discussion breaks down the attributes which have made some progress possible in quantification of water and air pollution aesthetic effects.

A major difference between water pollution and pesticide residue caused aesthetic effects is the possibility of assessing the source. Water pollution is commonly associated with major point sources. These sources are present over a long period of time and the fact of their existence becomes common knowledge. As pollution becomes apparent, these sources are identified and the amount of effluent discharged is measured. To the extent that the water district can do so, pressure can be brought to bear on the polluters to reverse the aesthetic damages. This is not to say, however, that in some cases non-point sources may be the major cause of pollutant loading (e.g., agricultural operations, forests, urban areas, etc.).

The fact that the aesthetic damages of water pollution can impact the viewers directly is what makes attempts at quantification somewhat successful. Water pollution is readily identifiable. Fish kills may become common, scum may float on the surface, vegetation may grow out of control, foul smell may emanate from the river. All of this makes human activity less enjoyable. Swimming and picnicking are curtailed. Tihansky (10) has made estimates of the loss to consumers of swimming at public beaches due to the closing of the beaches for health reasons. To do this, he drew on the fact that people were willing to pay for improved swimming conditions plus the unstated fact that closed beaches could be considered a homogenous entity for the purpose of the study. Further, it is reasonable to assume that the major component of the aesthetic loss was due to reduced swimming opportunities. As a first estimate of the aesthetic cost of water pollution, the study did succeed.

Finally, the impact of an increased dose of water pollution is predictable. The increased discharge of untreated waste into the waterway can only further degrade the aesthetic impression for an observer.

Measurement of the aesthetic effects of airborne pollution is just beginning. The sources of urban air pollution (a typical non-point source) can be generally specified into broad categories (transportation related, production related, and residentially related) but identification of individual air polluters and quantification of their contribution to the air pollution problem is difficult. Some of the large polluters can be identified, however, and if their contribution is a large proportion of the exhaust load, clean-up pressure can be brought to bear. Air quality indexes have been constructed and are in use in major cities. This serves as a warning device in the present and as an indicator of air quality trends in the long run. The aesthetic component of air pollution -- dirt, soot, odor and obscured views -- is real enough but difficult to measure in the urban environment due to the gradual nature of the pollution levels. With a continuous monitoring system this problem may be overcome and assessment such as took place in New Mexico may be possible.

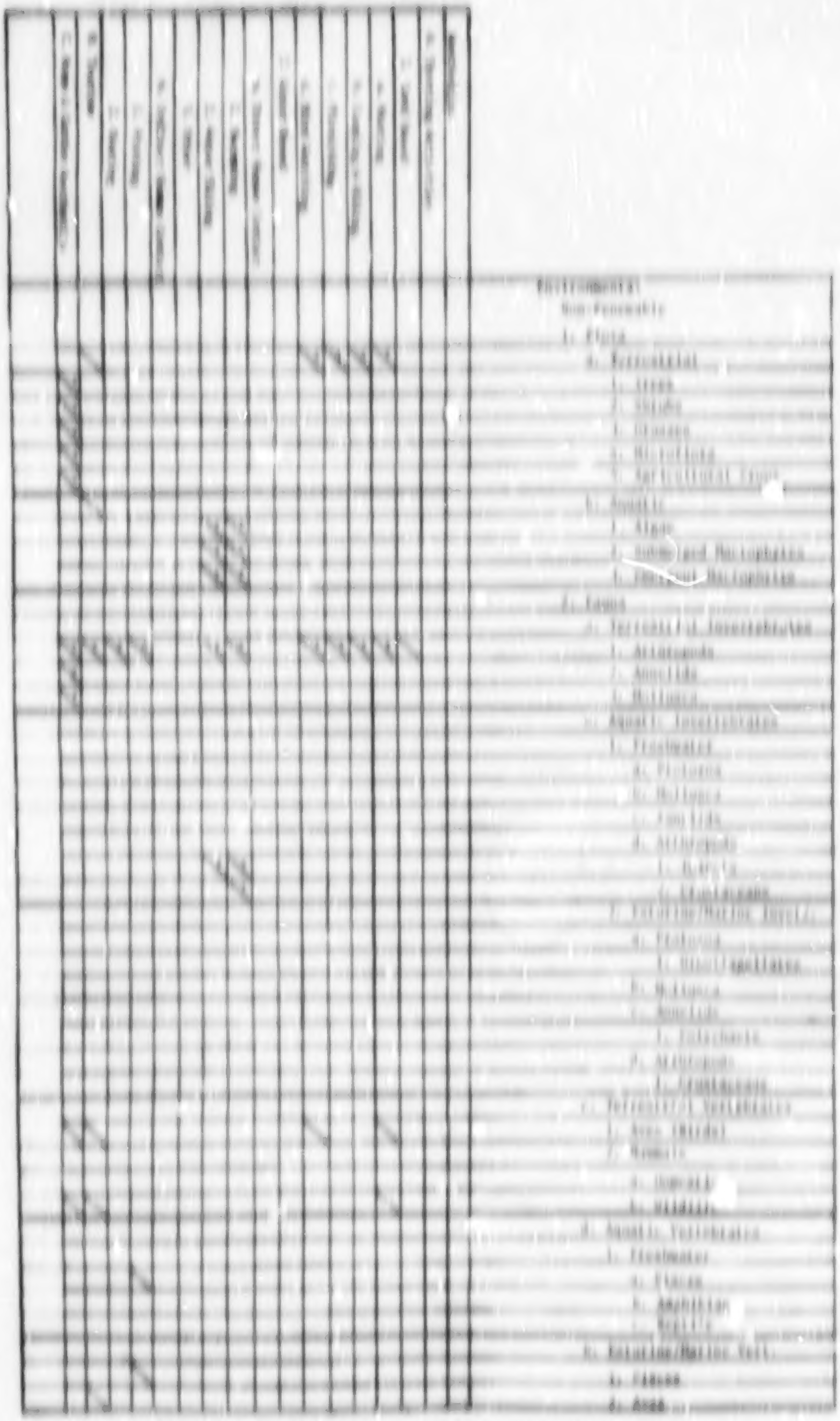
In the Four Corners area of New Mexico, Arizona, Colorado and Utah, a major source of air pollution was introduced. This coal burning power plant caused an aesthetic impact on an area whose major attribute was visual splendor. Randell *et al.* (8) undertook a survey of people's willingness to pay for restoration of the lost aesthetic opportunity and found significant support for clean air. The study was possible because the degradation was due to a clearly defined entity which was making an impact directly upon the individual.

AESTHETIC IMPACT ASSESSMENT

The state-of-the-art measurement techniques are by no means adequate vehicles for the quantification of pesticide impacts on aesthetic considerations. The majority of evaluatory schemes are site-specific since they were developed for air and water pollution and construction projects and are not readily adapted for the examination of aesthetic impacts of pesticide use. A number of assessment procedures have been critiqued by Redding (9).

In an attempt to insure that as few aesthetic consequences as possible are overlooked, we again propose the implementation of a checklist or matrix evaluation technique. The rationale for an aesthetic matrix is essentially the same as for an environmental effects matrix in that existing evaluation schemes do not readily permit quantification of environmental and aesthetic impacts. The matrix is designed in such a manner so as to insure that the evaluator will at least be aware of possible aesthetic ramifications. Figure 16 illustrates the envisioned aesthetic impact matrix.

The abscissa is comprised of the various components of the environment which may interact with the ordinate which is comprised of activities with aesthetic components. If an impact occurs, a slash is placed at the intersection of the two categories being examined. When attempting to assess the potential impacts (both beneficial and harmful) of a pesticide, using a matrix of this type would be inadequate unless coupled with an examination of food chain links and other environmental interrelationships to insure the consideration of not only first-round effects but also second- and third-round consequences of chemical compounds in the ecosystem. It may prove advantageous to implement a scheme consisting of a series of matrices examining first-, second-, and third-round effects. For example, an algacide may improve boating and fishing by reducing the amount of algae in the water (a "+" indicating a beneficial impact) but, may at the same time adversely affect fishing by having harmful effects on aquatic invertebrates and aquatic vertebrates. Pesticide use can affect the aesthetic component of hunting if the trees, shrubs and other terrestrial flora are adversely impacted. Conversely, if a pesticide



Environmental:
Non-Renewable
1. Flora
a. Terrestrial
1. Trees
2. Shrubs
3. Grasses
4. Microflora
5. Agricultural Crops
b. Aquatic
1. Algae
2. Submerged Macrophytes
3. Emergent Macrophytes
2. Fauna
a. Terrestrial Invertebrates
1. Arthropods
2. Annelida
3. Mollusca
b. Aquatic Invertebrates
1. Freshwater
a. Protozoa
b. Mollusca
c. Annelida
d. Arthropoda
1. Insects
2. Crustaceans
2. Estuarine/Marine Invert.
a. Protozoa
1. Dinoflagellates
b. Mollusca
c. Annelida
1. Polychaete
d. Arthropoda
1. Crustaceans
c. Terrestrial Vertebrates
1. Aves (Birds)
2. Mammals
a. Domestic
b. Wildlife
d. Aquatic Vertebrates
1. Freshwater
a. Pisces
b. Amphibian
c. Reptile
2. Estuarine/Marine Vert.
a. Pisces
b. Aves

Figure [illegible]. Pesticide impact on activities with aesthetic components reduces the number of insect pests in an area, this may improve the aesthetics of hunting. This insect-aesthetic relationship holds true with the remainder of the sporting activities and tourism. Pesticide-use decisions will, of course, severely impact upon hunting if there is an adverse effect on wildlife.

Assume that an insecticide is applied to control insects in the home and garden. At the onset, this action will have a positive impact on trees, shrubs, ornamental, etc. in that it kills arthropods (insects) and molluscs (slugs and snails). On the other hand, the pesticide may directly or indirectly kill annelids (earthworms) that are important to soil fertility and bees that are important in pollination. Beneficial birds and mammals may also succumb to the pesticide. Thus, an action to protect flora (e.g., garden crops) may result in production losses and pose serious environmental consequences.

The comprehensiveness of the analysis will depend upon the amount of available information on the particular pesticide, organisms and activities being considered. When considering effects other than first-round impacts, the analysis can continue *ad infinitum*. In this case the evaluator must set the limits of analysis.

THE ALDRIN CASE

In dealing specifically with aldrin application on corn, the information on direct aesthetic impacts is meager at best. This is most likely a result of tempering intermediates -- plant or animal -- tending to blur the causal chain between aldrin-dieldrin and environmental and aesthetic impacts. Environmental and aesthetic impacts must be considered together since an aesthetic impact is a consequence of an environmental alteration.

Though we are not able to establish a direct aesthetic impact related to aldrin use on corn, a cause and effect relationship has been drawn between dieldrin application on rice and the death of snow geese which fed on the treated rice. Therefore, the aesthetic impact of dieldrin can be alluded to since the death of snow geese will affect hunting and possibly landscape aesthetics. It is quite likely that incidents of this type are more common than reflected in the literature since scavengers tend to remove the carcasses very quickly and many birds may fall into cover.

The aesthetic impact of pesticides will most likely be less if they are used in a production sense -- to protect corn from insect damage -- than if used for aesthetic purposes at the onset (i.e., gypsy moth control in Northeastern forests). In any event, the causal chain must be established prior to aesthetic impact assessment since "natural death" is not as obtrusive to people as one attributed to preventable causes. That is, people will tend to assess the death of an organism differently if it is thought to have died of natural causes rather than the victim of pesticide poisoning. Therefore, the establishment of the correct cause and effect relationship surrounding aesthetic impacts is imperative if the assessment of such effects is to be correct.

BIBLIOGRAPHY FOR AESTHETIC EFFECTS CHAPTER

1. Birkhoff, G. D. Aesthetic Measure. Harvard University Press, Cambridge, Mass., 1933.

2. Coomber, N. H. and A. K. Biswas. Evaluation of Environmental Intangibles. Genera Press, Bronxville, New York, 1973.

3. Diffey, T. J. Evaluation and Aesthetic Appraisals. British Journal of Aesthetics, 7(4): 358-373, 1967.

4. Grimes, O. F. Evaluation of Recreation and Aesthetic Uses of Water in an Urban Setting. Paper presented at the Urban Economics Workshop, University of Chicago, February, 1970.

5. Knopp, T. B. Environmental Determinants of Recreation Behavior. Journal of Leisure Research, 4(2): 129-138, 1972.

6. Leopold, L. B. Method for Measuring Landscape Appeal. Natural History, 8: 36-45, October 1969.

7. Peebles, J. J. A Methodology Study to Develop Evaluation Criteria for Wild and Scenic Rivers. Report of Flood Control Subproject, Water Resources Institute, University of Idaho, February 1970.

8. Randall, A. B., *et al.* Benefits of Abating Aesthetic Environmental Damage from the Four Corners Power Plant. New Mexico State University Agricultural Stat. Bulletin, Fruitland, New Mexico, 1974.

9. Redding, M. J. Aesthetics in Environmental Planning. U. S. Environmental Protection Agency, Washington, D. C., 1973.

10. Tihansky, D. P. Recreational Welfare Losses from Water Pollution Along U. S. Coasts. Journal of Environmental Quality, 3(4): 335-342, 1974.

CHAPTER 7

DISTRIBUTION EFFECTS*

The preceding sections have emphasized the quantifiable benefits and costs as well as some descriptions of the effects of pesticide decisions on the health of humans and non-humans. In this section, procedures will be described for analyzing and presenting the effects of pesticide-use decisions that relate to the distribution of benefits and costs. Specific attention is directed to the impact on geographic distribution, social effects, and effects on the national balance of payments.

GEOGRAPHIC DISTRIBUTION OF EFFECTS

The geographic distribution of effects from the pesticide-use decision comes primarily from models of economic production effects and from the dispersion of human health effects. These models will permit the determination of the balance of benefits and costs by geographic region. Appropriate regions will be determined by the particular pesticide decision under investigation. For some, the geographic areas may be rather small and carefully delineated. In other decisions, the geographic area may encompass large portions of the nation such as the Southeast, the Northeast, and the West. The requirements of the particular pesticide decision determine the appropriate geographic areas for each report.

One of the most likely geographic effects from a pesticide-use decision is the relocation of agricultural production. As was shown in Chapter 3 in the example of aldrin used on corn, even when total production is virtually constant, certain regions may have significant increases or decreases in the production of a particular crop which can lead to further important changes in the economy of the area. If corn production, for example, is significantly reduced in an area, the demand for production inputs, such as fertilizer and specialized corn production machinery, will be reduced. This may lead to reduced employment in farm supply firms and perhaps the closing of some firms. In addition, there will be less corn to be stored or shipped by grain handling companies; again leading to reduced employment in the area. If the reduction in corn production continues, it is likely that livestock feeding activities will be curtailed or discontinued, reducing the employment in livestock shipping companies, marketing firms (e.g., auction markets) and packing plants.

---

* Principal researcher in this section is Dr. Kenneth P. Wilkinson, Department of Rural Sociology, The Pennsylvania State University. This section was prepared under the supervision of Dr. Donald J. Epp, Principal Investigator.

In evaluating these possible changes, the analyst must consider the availability of other employment in the region, the magnitude of the lost employment relative to total area employment and linkages in the local economy which may accentuate or mitigate the impact of agricultural production and employment shifts.

A similar consideration should be given to areas where agricultural production is projected to increase. A substantial increase in soybean production, for example, may lead to changes in farm size and, therefore, the number of farm families. New businesses, such as feed supplement manufacturing, may come into the area bringing increased employment and possibly an inflow of new people to the area. Again, a knowledge of linkages in the local economy will help the analyst predict the economic ramifications of these changes.

Techniques for measuring linkages in the regional economy and developing employment and income multipliers are available. Isard (4) describes how these measures are calculated and possible extensions for analyzing regional effects of a change in basic employment and production. Input-output analysis and the resulting multipliers must be used with caution, however, since they assume that the basic structure of the economy remains constant. If a pesticide registration decision causes a major change in the agricultural production of a region, the multipliers calculated from the previous economic structure are likely to be wrong.

Significant human health effects or changes in the environment can also differ from region to region and may lead to economic and social changes. For instance, the denuding of large areas of forest land, due to discontinued spraying of insects, may lead to a reduction in tourist visits to the area and a decrease in the value of seasonal home sites. These changes could cause loss of seasonal employment and the failure of some tourist oriented businesses, a reduction in sales tax and real estate tax revenues and a reduction in the quantity or quality of local governmentally provided services. This could accelerate the decline in local population. On the other hand, an area that discontinues the use of potentially harmful pesticides may experience an inflow of people who appreciate the "safer" environment or the wildlife that is more abundant without the adverse chemical effects.

A more complete analysis of region impacts will include the likely social ramifications of altered economic activity. This analysis is described in the following section.

SOCIAL EFFECTS

Social effects of any policy decision are difficult to foretell, because of the complexity of forces involved, and difficult, if not impossible, to evaluate on objective grounds. Attempts to bring about specifically desired social changes, particularly in complex social systems, rarely succeed. Indeed, as Jay Forrester (3) among others has pointed out, the complex web of "feedback loops" in a social system produces such an array of unanticipated consequences that well-intended policy action often results in changes in direct opposition to those desired. For a simple example, an economic development program intended to reduce poverty in a community might produce, in order, new jobs, an influx of poorly trained migrants seeking service employment, resulting in more poverty. Breaking into such cycles with any chance of successful policy impact, requires, as a minimum requisite, a sense of the complexity of social systems. Social changes are difficult to evaluate objectively because of the inherently subjective character of evaluation criteria. A given change, for example, from a pattern of family farms to one of vertically linked corporate structures, might be viewed as desirable from one value perspective and undesirable from another. There is no universal basis, grounded in scientific knowledge, for assessing the relative goodness of any value system; rather value systems must be treated as integral wholes each with its strengths and weaknesses for meeting human needs. Among social scientists interested in policy effects, one response to problems of complexity and value relatively in social impact analysis has been to attempt to develop social accounting systems rather than programmic, goal-oriented models of social change.

A Social Accounting System

A social accounting system is based on a taxonomy of aspects of social organization likely to be affected by a policy decision. At this stage of the development of social science, such a taxonomy must be conceptually, rather than empirically generated. Among alternative theories of social organization there is some consensus as to the basic elements (but much disagreement as to how these elements operate and interrelate in the functioning and change of social systems). Abstracting commonalities from diverse theoretical approaches reveals the following categories of elements of social organization which must be considered in any social impact assessment:

- Human Ecology: Spatial distribution and interrelationships of people in adaptation to the natural environment (e.g., population size, composition and change; sustenance organization; technology).

- Culture: Values and norms, particularly institutions, in social life (e.g., value-based norms governing family life, economic activities, education, religious life, the polity, etc.).

- Social Structure: Group relationships expressing cultural values and norms (e.g., families, voluntary organizations, firms, friendship groups, government agencies, social classes, social position and role networks).

- Social Processes: Interaction dynamics (e.g., cooperation, competition, conflict or the dynamics of social structure).

- Social Psychology: Relation of the individual to society (e.g., participation and ego-involvement, self-concept and reference-identification, "mental health").

The question of how these elements of social organization might be affected by a policy change, such as regulation of a given pesticide, should be approached with consideration of the following issues:

Social Significance--

Of what social significance (in terms of the above categories) is the policy variable (e.g., the pesticide practice) in the relevant unit or level of social organization (community, region, nation, etc.)? A crude approach to answering this question would be simply to enumerate the number of elements of social organization likely to be affected directly and obviously by a policy change. Even a gross estimate, however, would have to be couched in terms of a description of the major elements of current social organization.

Expected Changes in Social Organization--

What kinds of changes should be expected in those aspects of social organization likely to be affected directly and obviously by the policy change, and what subsequent changes should be expected in turn? This, of course, raises again the issue of complexity in social forecasting. Two sources of guidance, neither being in any sense definitive, are available in the sociological literature. One is in the theories of social change which have been developing over many years and the other is in the body of empirical findings of sociological research, much of which dates only to the past few decades.

Theories of social change are many and competitive. For convenience these have been grouped, more or less satisfactorily in general treatments, in two main types according to philosophic premises upon which they are based. One type of theory of social change is based on the assumption that social life is governed by a set of principles of order which prevade all aspects of existence in the universe. An example is the theory referred to as functionalism, which assumes that changes in any part of a social system set off complementary changes in other parts so as to retain the fundamental equilibrium of the whole system. A second type of theory of social change is based on the assumption that order is created by the acts of men, that, in fact, the natural direction of change in all elements of the universe is toward random dispersion or entropy. Conflict theory, for example, assumes that change occurs as the result of the confrontation of opposed interests of independent groups. Interactional theory, a second example of the latter type, assumes that social life reflects the collective interests of people rather than the functional needs of systems. Empirical testing of propositions derived from these competing theories has proceeded at a very slow rate, and is in fact at an early stage.

The relevant body of empirical evidence relates to what has been called "theories of the middle range," that is, to theoretical propositions which do not relate explicitly to the more global, philosophic assertions. A number of bivariate relationships, such as that between social participation and socio-economic status, have been reported with such frequency in sociological studies as to give them the status of middle-range principles. The testing of multivariate models is a recent innovation in sociological research.

Desirability of Expected Changes--

Are the changes anticipated desirable? This of course is a value question, for which there is no value-free answer. Once certain values have been accepted, however, weights and directions can be assigned to the changes which are anticipated. A number of observers have attempted to ascertain the fundamental values pursued in American Society. Central to most treatments are the following:

- Efficiency (especially in adaptive or economic activities).
- Equity (especially in distribution of benefits and costs of change).
- Community (especially in the sense of integration of behavior and identities.)
- Self-actualization (i.e., the fulfillment of distinctively human potentialities--which apparently requires at least minimal levels of efficiency, equity and community).

Community Level Assessment

The most relevant unit for assessment of social effects of pesticide regulation decisions is the local society or community. For larger units, such as the nation, economic, environmental and health consequences would be more clearly discernible. Social effects at the national level would be diverse and, perhaps, counter-balancing among themselves. One exception at the national level would be in the effects on concentration of power in the control of agriculture. While family-type enterprises continue to be the unit in the agricultural sector, large-scale, vertically linked corporate structures account for much of the growth in agricultural production in recent years. In general, larger corporate structures should be expected to have greater resources to apply in adapting to a regulatory decision; thus further growth of such structures--with obvious ramifications in social organizations might be encouraged by such decisions. However, the forces encouraging the development of such structures are much broader than those having to do with use of specific pesticides. Increased pesticide use and complex organizational structure are part of a process, sometimes referred to as modernization, which has been occurring in accelerated form in virtually all sectors of Western society for many years. It is highly unlikely that any action by government, or any set of purposive actions, could alter the fundamental direction of this process. Social effects which could be effectively monitored and perhaps affected are best seen in the variety of circumstances prevailing at the local level.

In describing effects at the local level, it is important to have a realistic concept of the geographic scope of the local society. Areas which were relatively independent communities in former times might now be sub-units of a larger community. Sociological studies of communities said to be "declining," for instance, often reveal that the community is in fact expanding as services and institutional functions formerly associated with a specific, limited place move to a larger regional center. The general criterion for operational definition of a local society is the presence in it of the full range of institutional forms available in the larger society. Functional economic area delineations often provide an adequate starting place for identifying community boundaries.

At the community level, social effects of the regulation of a given pesticide will depend in part on the characteristics of the community and the significance of the use of the pesticide in its structure and functioning. Considerable work has been done in recent years by Brian Berry (2) and other central-place geographers in the use of factor analysis and secondary data sources to classify community types. Such work might be used to estimate the significance of agriculture in community life. The general indication of research in this area is that the number of communities organized primarily around agriculture has declined significantly since World War II and is likely to decline further. The number of such communities in which a given pesticide is of crucial significance in agriculture is small, though still large enough in absolute terms to warrant national concern for social consequences of regulation. In most other communities, i.e., those in which agriculture is of relatively minor significance or in which a given pesticide is not crucial, the social effects of a change in policy are likely to be slight, although they might be intensively experienced by some individuals and families.

It is the case of the clear significance of pesticide use in community life that the question as to what effects are likely becomes most relevant. If, for example, regulation means that the type of agricultural enterprise in the area must be altered, or replaced with non-agricultural activities, the effects could be far-reaching. The size and age-composition of the population might be altered, resulting in a new mix of orientations, needs and demands in community affairs. Tax revenues for support of public services and amenities might be curtailed. Status relationships might be altered. Linkages to the larger society through formal agencies might be affected. The specific areas of likely impact would depend upon characteristics of the given local situation.

The "state of the art" and the "nature of the beast" are such that social effects cannot be foretold with precision. The theories, methods and findings of sociology could be used, however, to identify significant issues and social parameters to be considered in the making of policy decisions.

BALANCE OF PAYMENTS EFFECTS

The purpose of this section is to investigate the impact that a pesticide-use decision might have on the international trade position of the

United States. This section does not contribute to a benefit-cost ratio directly but is considered under the heading of distribution. As such, the results of this section are considered as auxiliary information available to the decision-maker.

Impact of Foreign Trade

Foreign trade allows a country to improve its efficiency by purchasing from abroad those products which can be produced more efficiently abroad. Payment for these products must ultimately be made by sales of domestic goods to foreigners. It can be said that a country exports in order to pay for its imports.

U.S. agriculture makes a significant contribution to a positive trade balance. In fiscal year 1975, the value of agricultural exports was 21 percent of the total value of U.S. exports. The value of agricultural imports was 9 percent of the total value of U.S. imports. This resulted in a positive agricultural trade balance of over $12 billion which was able to cover a negative non-agricultural trade balance of over $1.8 billion. Table 15 shows that the export value of just seven crops accounted for over 71 percent of total U.S. agricultural export value in fiscal year 1974 and 1975. Percentage of total domestic crop which entered into foreign trade is given in Table 16. These tables demonstrate the importance of major agricultural products to U.S. trade balance.

In Chapter 3 and in U.S. Environmental Protection Agency (7), the quantitative analysis of production effects of restricting the use of aldrin on corn assumed that any changes in production caused by a decision in pesticide use would not significantly affect the quantity of product consumed domestically. It is possible, however, that a relatively small change in the amount produced may have a significant effect on the amount of produce exported. The difference in effect on domestic and foreign markets is primarily due to the greater price elasticity of demand in the foreign market than in the domestic market for most products dependent on pesticides for their production. Another factor which may reduce the amount of export sales, particularly of agricultural products, is that a reduced supply limits the opportunities for special sale efforts tied to foreign policy considerations. Both of these factors, a slightly increased price and reduced foreign policy arrangements, may reduce the amount of foreign exchange earnings from the sale of agricultural products.

The method for calculating the foreign exchange effects utilizes price changes from the simulation model in the economic effects section along with estimates of the elasticity of demand in the foreign market. These calculations indicate changes in foreign demand for the products under consideration and iterate with the simulation model through adjustment of the exogenous foreign demand components that are added to various regional demand quantities. If it appears that this alters the results of the simulation analysis, the procedure is rerun using the new foreign demand quantities. After a final foreign demand effect is calculated, it is translated through current prices or projected prices into a foreign exchange earnings adjustment.

TABLE 15. MAJOR U.S. EXPORT CROPS

1974 - 1975

Crop	Fiscal Year 1974[1]	Fiscal Year 1975[2]
Wheat	$ 4,556,405,000	$ 4,797,015,000
Corn	3,729,142,000	3,988,207,000
Soybeans	3,272,832,000	2,951,232,000
Cotton	1,293,999,000	1,016,953,000
Tobacco	813,601,000	910,088,000
Rice	752,282,000	1,002,185,000
Grain sorghums	631,127,000	616,308,000
	$15,049,388,000	$15,281,988,000
Total U.S. Agricultural Exports	$21,293,000,000	$21,584,000,000

SOURCES: 1) U.S. Foreign Agricultural Trade Statistical Report, Fiscal Year 1974. Economic Research Service, U.S. Department of Agriculture, October 1974.

2) U.S. Foreign Agricultural Trade Statistical Report, Fiscal Year 1975. Economic Research Service, U.S. Department of Agriculture, December 1975.

TABLE 16. UNITED STATES EXPORTS OF MAJOR CROPS AS A PERCENTAGE OF TOTAL UNITED STATES PRODUCTION

Crops	Quantity Exported[1] (Fiscal Year 1975)	Domestic Production[2] (1974)	Percent
Wheat	999,230,000 bu.	1,796,187,000 bu.	55%
Corn	1,122,137,000 bu.	4,663,631,000 bu.	24%
Soybeans	404,514,000 bu.	1,214,802,000 bu.	33%
Cotton	3,816,000 bales	11,540,100 bales	33%
Tobacco	638,404,000 lbs.	1,989,720,000 lbs.	32%
Rice	5,059,180,000 lbs.	11,230,400,000 lbs.	45%
Grain sorghums	190,894,000 bu.	629,222,000 bu.	30%

SOURCE: 1) U.S. Foreign Agricultural Trade Statistical Report, Fiscal Year 1975. Economic Research Service, U.S. Department of Agriculture, December 1975.

2) Crop Production. Statistical Reporting Service, U.S. Department of Agriculture, January 15, 1976.

It is entirely possible that a pesticide decision may have a very minor effect on total production of a particular crop and an insignificant effect on domestic price but at the same time cause a substantial adjustment in foreign sales and have a significant effect on foreign exchange earnings. This could be an important consideration in making a decision since agricultural exports recently have accounted for large portions of the foreign exchange earnings of the United States.

The Effects on the U.S. Trade Balance of Restricting Aldrin on Corn

Corn is used throughout the world as a feed grain for cattle, hogs and poultry. A large quantity is also used for human consumption. The U.S. contribution to this worldwide demand for corn is substantial and the return to the U.S. in terms of foreign currency earnings is likewise substantial. As shown in Tables 15 and 16, the U.S. exported approximately 24 percent of its 1974 corn crop and received nearly $4 billion from this sale. The magnitude of this dollar flow indicates that the result of even a small percentage change in output can be a significant change in the amount of absolute dollar terms. It is for this reason that the impact on the U.S. trade balance of a change in the production function of corn (due to the restriction of aldrin on corn) is examined.

As described in the previous section, the procedure to be followed in this evaluation of foreign exchange effects is to estimate the domestic price change expected from the registration action, and, using estimates of the elasticity of foreign import demand, calculate the new quantity demanded by foreign buyers. The domestic price change is derived from the simulation model described in Chapter 3. The estimate of foreign import demand is obtained from a U.S.D.A. simulation model titled *Commodity Projections for 1985* (6).

Among the commodities which are entered into the U.S.D.A. model are wheat, coarse grains (corn, barley, oats, sorghum), oil seeds, beef and dairy products. This model breaks the world into developed and less-developed areas since it has been found that these two groupings have significantly different reactions to price and quantity changes. The estimate of the elasticity of foreign import demand for corn obtained from this model is .3 for the developed area and .65 for the less-developed. This means that for a 10 percent increase in the price of corn, foreigners from developed countries will demand a 3 percent lower quantity and foreigners in less-developed countries will demand a 6.5 percent lower quantity. Both of these elasticity estimates are inelastic.

The results of the production-section modeling indicate that if aldrin is restricted from use on corn soil insects, the response of the farm sector is one of increasing the acreage devoted to corn as well as some shifts in cropping location. This results in a maintainence of the quantity of corn produced and a stable price of corn. With no change in domestic price, there can be no predicted change in the quantity exported due to a domestic price change.

BIBLIOGRAPHY FOR DISTRIBUTION EFFECTS CHAPTER

1. Abel, Martin E. and Frederick V. Waugh. Measuring Changes in International Trade. Journal of Farm Economics, 48, Nov. 1966.

2. Berry, Brian, ed. City Classification Handbook. Wiley, New York, 1972.

3. Forrester, Jay W. World Dynamics. Wright-Allen Press, Cambridge, Mass., 1971.

4. Isard, Walter. Methods of Regional Analysis: An Introduction to Regional Science. MIT Press, Cambridge, Mass., 1960.

5. Rojko, Anthony S., Francis S. Urban and James J. Native. World Demand Prospects for Grain in 1980. Foregin Agricultural Economic Report No. 75, USDA, ERS, Dec. 1971.

6. U.S. Department of Agriculture. Commodity Projections for 1985. Economic Research Service, undated.

7. U.S. Environmental Protection Agency, Economic Analysis Branch, Criteria and Evaluation Division, Office of Pesticide Programs. Detailed Information on Linear Programming Analysis of Organochlorine Suspension of Corn Use in the United States. September 1975.

8. Waugh, Frederick V. Demand and Price Analysis, Some Examples from Agriculture. Technical Bulletin 1316, USDA, ERS, Nov. 1964.